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Conflicts of human land-use and conservation areas: The case of Asian elephants in rubber-dominated landscapes of Southeast Asia

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Author's Declaration

I, Franziska Kerstin Harich, hereby affirm that I have written this thesis entitled "Conflicts of human land-use and conservation areas: The case of Asian elephants in rubber-dominated landscapes of Southeast Asia" independently as my original work as part of my dissertation at the Faculty of Agricultural Sciences at the University of Hohenheim.

All authors in the quoted or mentioned publications in this manuscript have been accredited. No piece of work by any person has been included without the author being cited, nor have I enlisted the assistance of commercial promotion agencies. This thesis has not been presented into other boards for examination.

Stuttgart, 2017

Franziska Kerstin Harich

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Summary

Over the last decades, expanding rubber plantations in Southeast Asia have continuously diminished natural wildlife habitat, thereby increasing conflicts between human land-uses and nature conservation. The consequences of the rubber boom are manifold, with short-term economic benefits for smallholder farmers and long-term costs for species diversity and ecosystem services (ESS). Sustainable wildlife populations are critical for ecosystem functioning but the ongoing habitat degradation and conflicts with people threaten the survival of larger mammal populations. This trend is particularly problematic if the respective species in decline are keystone species such as the Asian elephant (*Elephas maximus*), which holds important ecological functions in maintaining tree diversity.

Continuous land-use transformations increase the importance of conservation efforts for wild mammal diversity within the agricultural matrix surrounding natural areas. In order to develop a profound knowledge basis for effective protection strategies, the major aim of this thesis' work, therefore, was to analyze the potential of rubber-dominated landscapes in sustaining wildlife populations while considering the risk of conflicts due to wildlife damage. Furthermore, the ecological importance of large mammals was assessed using the example of Asian elephants. In addition, a methodology was developed to incorporate multiple levels of species diversity into ESS evaluation models.

As a first step, the available literature on mammalian wildlife in rubber plantations was analyzed to provide an overview on species diversity found in these land-use systems on a large geographical scale. This provided indications for the potential of wildlife-friendly farming approaches. As the available data on wildlife diversity in rubber cultivation was scarce, oil palm plantations were included as a reference for general ecological impacts of tropical tree crop systems on animal populations. Our review showed that species richness was highly reduced in the plantations compared to natural forest and that most species in rubber and oil palm farms were rather visitors than permanent residents.

For ground-truthing and a more detailed assessment of species richness and presence in rubber plantations, transect and camera trap surveys were conducted in the farm-forest transition zone of the Tai Rom Yen National Park, located in a typical rubber-dominated landscape in southern Thailand. Furthermore, farmers were interviewed on the kind and extent of wildlife damage and their perceptions towards nature conservation. With 35 recorded wildlife species, the forest was found to hold the highest diversity while more

than 70% of these mammals were still found at the forest edge. However, a strong decline of species diversity and presence was observed in the surrounding farmland. Crop damage by wildlife affected 40% of all interviewed farmers. In 85% of all rubber damage incidents, young trees were affected, which had not yet been tapped. Elephants were most frequently named as damage causing species. Nevertheless, damage to rubber occurred only in half of the elephant visits, indicating that these crop species were not particularly attractive to foraging wildlife.

To account not only for the costs inflicted through elephant damage but also for the ecological benefits elephants provide, the potential of these megaherbivores for seed dispersal was assessed as a crucial ecological function in forest ecosystems. Feeding experiments with captive elephants were conducted and germination success of ingested and fresh control seeds of a tree species with characteristic mega-faunal syndrome fruits (*Dillenia indica* L.) was monitored. Seeds that were ingested by elephants showed a significantly higher and earlier likelihood for germination compared to control seeds. The exemplary tree species in our experiments did not solely depend on but benefited from elephant consumption for germination. This highlights the risks of long-term negative implications for certain tree species and entire ecosystems if elephant and other large mammal populations further decline.

Biodiversity including mammalian wildlife diversity is an integral and essential component of ecosystem functioning and the provisioning of services. However, a challenge in the evaluation of ESS is the allowance for the multiple facets of biodiversity assessments. We therefore developed a methodology for including multiple levels of species diversity into an integrated ESS evaluation model. Diversity data of mammals, birds, reptiles, amphibians, insects and plants derived from our multi-disciplinary data collections and from literature sources were normalized using the most diverse habitat as benchmark. Through this approach we obtained a comparable habitat suitability matrix for different land-use systems, which was then applied to different future land-use scenarios. The outcomes confirmed that a conservation focused scenario scored higher habitat suitability for all species as well as for threatened ones compared to two other scenarios with no or limited conservation measures.

Increasing conflicts between human land-use and nature conservation as a result of shrinking resources pose imminent risks for the diversity and resilience of ecosystems. This thesis provides an assessment of the current state of and conflicts with wildlife diversity in rubber-dominated landscapes surrounding protected areas. The results of this thesis can serve as a basis for the development of measures to consolidate farming and conservation interests. The method of incorporating multiple species levels in ESS evaluation models further allows for assessing the impacts of different land-use management decisions, which can be transferred to other regions and land-use types as well. Although intensively managed plantations cannot substitute for natural forests, efforts are required to conserve multiple levels of biodiversity within the farming landscape. High species diversity will maintain ecosystem functions and services sustainably, which both human and wildlife communities rely on for their long-term persistence.

Zusammenfassung

In den letzten Jahrzehnten wurde der Kautschukanbau in Südostasien kontinuierlich ausgeweitet, was zu zunehmenden Konflikten zwischen menschlicher Landnutzung und Naturschutz beigetragen hat. Die Konsequenzen des Kautschukbooms sind vielfältig und beinhalten sowohl kurzfristige wirtschaftliche Vorteile für kleinbäuerliche Betriebe als auch nachhaltige Kosten für die Artenvielfalt und für Ökosystemdienstleistungen. Wildtierpopulationen nehmen essenzielle Funktionen in Ökosystemen ein, doch die zunehmende Zerstörung von natürlichem Lebensraum bedroht das Überleben vieler Säugetierpopulationen. Diese Entwicklung ist vor allem dann problematisch, wenn die betreffende rückläufige Art eine sogenannte „Schlüsselart" ist, welche wichtige ökologische Aufgaben übernimmt. Der asiatische Elefant (*Elephas maximus*) ist solch eine Schlüsselart, die zur Erhaltung der Vegetationsvielfalt in natürlichen Wäldern beiträgt.

Als Folge des fortdauernden Landnutzungswandels gewinnen Maßnahmen zum Schutz von wilden Säugetieren in der Agrarlandschaft an immer größerer Bedeutung, vor allem in der Nähe zu natürlichen Wäldern. Voraussetzung für die Etablierung effektiver Schutzmaßnahmen ist dabei eine fundierte Wissensbasis. Das Hauptziel dieser Dissertation war deshalb, das Potenzial von kautschukdominierten Landschaften zum Erhalt von Wildtierpopulationen unter Berücksichtigung von Konfliktrisiken durch Wildtierschäden zu analysieren. Anhand des Beispiels von asiatischen Elefanten wurde zudem die ökologische Bedeutung von wilden Großsäugern untersucht. Weiterhin wurde eine Methode entwickelt, um verschiedene Ebenen der Artenvielfalt in ein Model zur Analyse von Ökosystemdienstleistungen einzubeziehen.

In einem ersten Schritt wurde die verfügbare Literatur zu wilden Säugetieren in Kautschukplantagen ausgewertet um einen Überblick zur Artenvielfalt in diesen Landnutzungssystemen auf breiterer geografischer Ebene zu erhalten. Hierdurch wurden Indikatoren für das Potenzial von nachhaltigeren Anbaumaßnahmen gewonnen, welche zur Förderung von Wildtierpopulationen beitragen können. Da nur sehr begrenzt Daten zur Vielfalt von Wildtieren in Kautschukanbausystemen zur Verfügung standen, wurden Erkenntnisse aus Ölpalmplantagen einbezogen um Vergleiche zu generellen Umweltauswirkungen von tropischen Baumplantagen auf Tierpopulationen ziehen zu können. Die Begutachtung zeigte, dass die Artenvielfalt in den Plantagen im Vergleich zu

natürlichem Wald stark reduziert war und dass die meisten Arten in Kautschuk- und Ölpalmplantagen eher kurzfristige Besucher statt dauerhafter Bewohner waren.

Um diese Literaturergebnisse durch Felddaten zu verifizieren und detailliertere Kenntnisse zur Artenvielfalt und Artenpräsenz in Kautschukplantagen zu erarbeiten wurden Untersuchungen in der Übergangszone zwischen natürlichem Wald und landwirtschaftlicher Nutzfläche anhand sogenannter „Transekte“ und Kamerafallen durchgeführt. Bei dem Untersuchungsgebiet in und um den Tai Rom Yen Nationalpark handelte es sich um eine typische kautschukdominierte Landschaft im Süden von Thailand. Lokale Landwirte wurden zu der Art und dem Ausmaß von Wildtierschäden sowie zu ihrer Haltung gegenüber Naturschutz befragt. Mit insgesamt 35 dokumentierten Wildtierarten war der natürliche Wald am artenreichsten. Während immerhin noch mehr als 70% der Tierarten an der Waldgrenze gefunden wurden, war deren Anzahl und Präsenz in der Nutzlandschaft stark reduziert. Wildtierschäden betrafen 40% aller befragten Landwirte. In 85% der Fälle in denen Kautschukschäden auftraten waren junge Bäume betroffen, die noch nicht zur Ernte angezapft worden waren. Elefanten wurden am häufigsten als schadenverursachende Wildtierart genannt. Nichtsdestotrotz traten Schäden an Kautschukbäumen nur in der Hälfte der Fälle auf, in denen Elefanten die Plantagen aufsuchten. Daraus lässt sich ableiten, dass Kautschuk keine besonders attraktive Futterquelle für die Wildtiere darstellt.

Elefanten verursachen allerdings nicht nur Kosten, sondern liefern auch umfassende ökologische Nutzen. Um solche Vorteile zu berücksichtigen wurde das Potenzial der Megaherbivoren für die Samenverbreitung als Beispiel für wichtige ökologische Funktionen in Ökosystemen untersucht. Hierfür wurden Fütterungsexperimente mit domestizierten Elefanten durchgeführt und der Keimerfolg verfütterter Samen und frischer Kontrollsamen beobachtet. Bei den untersuchten Samen handelte es sich um *Dillenia indica* L., eine Baumart mit Früchten, welche typische Merkmale für das sogenannte „Megafauna Syndrom“ aufweisen. Samen von Früchten, die von Elefanten gefressen worden waren, wiesen eine signifikant höhere und frühere Keimwahrscheinlichkeit auf als entsprechende Kontrollsamen. Die exemplarische Baumart in unseren Experimenten war nicht allein auf Elefanten für die Keimung angewiesen, profitierte aber von dem Konsum durch die Tiere. Dies verdeutlicht die langfristigen Risiken für bestimmte Baumarten und

ganze Ökosysteme, falls die Elefantenpopulationen und andere Gruppen an Großsäugern weiter abnehmen.

Biodiversität, einschließlich der Säugetiervielfalt, ist eine integrale und essenzielle Komponente in Ökosystemfunktionen und den damit zusammenhängenden Dienstleistungen. Eine Herausforderung in der Evaluation dieser Services ist jedoch die Berücksichtigung der zahlreichen Facetten von Biodiversitätsanalysen. Aus diesem Grund wurde eine Methode entwickelt um verschiedenste Ebenen der Artenvielfalt in ein integriertes Evaluationsmodel für Ökosystemdienstleistungen einbeziehen zu können. Daten zu der Artenvielfalt von Säugetieren, Vögeln, Reptilien, Amphibien, Insekten und Pflanzen aus unseren multidisziplinären Projekten sowie aus Literaturquellen wurden normalisiert, wobei der jeweils artenreichste Lebensraum als Referenz diente. Durch diesen Ansatz wurde eine vergleichbare Matrix zur Habitatseignung von verschiedenen Landnutzungssystemen geschaffen, welche wiederum auf zukünftige Landnutzungsszenarien angewandt werden konnte. Die Ergebnisse bestätigten, dass ein auf Naturschutz ausgerichtetes Szenario für alle und insbesondere auch für gefährdete Arten eine höhere Eignung als Lebensraum erzielte als zwei andere Szenarien ohne bzw. mit begrenzten Naturschutzkomponenten.

Zunehmende Konflikte zwischen menschlicher Landnutzung und Naturschutz als Folge abnehmender Ressourcen stellen große Risiken für die Vielfalt und Stabilität von Ökosystemen dar. Diese Dissertation liefert eine Analyse des derzeitigen Zustandes der Artenvielfalt von Wildtieren sowie den Konflikten zwischen Mensch und Tier in kautschukdominierten Landschaften um Naturschutzgebiete. Die Ergebnisse dieser Arbeit können als Grundlage für die Entwicklung von Maßnahmen dienen um landwirtschaftliche und ökologische Interessen zu vereinbaren. Die Methode zur Einbeziehung verschiedener Ebenen der Artenvielfalt erlaubt es, die Auswirkungen unterschiedlicher Landnutzungsentscheidungen zu analysieren. Diese Vorgehensweise kann zudem auf andere Regionen und Landnutzungsarten angewandt werden. Obwohl intensiv bewirtschaftete Plantagen keine natürlichen Wälder ersetzen können, bedarf es dennoch Anstrengungen um mannigfaltige Formen von Biodiversität auf landwirtschaftlichen Nutzflächen zu erhalten. Eine hohe Artenvielfalt ermöglicht den nachhaltigen Schutz von Funktionen und Dienstleistungen von Ökosystemen, auf welche sowohl Menschen als auch Wildtiergemeinschaften langfristig angewiesen sind.

List of publications included in this thesis

In order to adhere to the regulations for a cumulative PhD thesis at the Faculty of Agricultural Sciences, several publications have been included into this work. These publications have been edited to fit the regulations of different peer-reviewed scientific journals as well as the overall style of this thesis. The references for all chapters are compiled at the end of this thesis.

Chapter 2:

Harich, F.K. & Treydte, A.C. (2016). Mammalian wildlife diversity in rubber and oil palm plantations. *CAB Reviews, 11* (020), 1-11. doi:10.1079/PAVSNNR201611020
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Chapter 4:

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Chapter 5:

Cotter, M., Häuser, I., Harich, F.K., He, P., Sauerborn, J., Treydte, A.C., Martin, K., Cadisch, G. (2017). Biodiversity and ecosystem services - A case study for the assessment of multiple species and functional diversity levels in a cultural landscape. *Ecological Indicators* 75, 111-117. doi:10.1016/j.ecolind.2016.11.038
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List of abbreviations and acronyms

DNP	Department of National Parks, Wildlife and Plant Conservation
ESF	Ecosystem Function(s)
ESS	Ecosystem Service(s)
HWC	Human-wildlife conflict
InVEst	Integrated Valuation of Ecosystem Services and Trade-offs
IUCN	International Union for Conservation of Nature
LDD	Land Development Department Thailand
NP	National Park
NRCT	National Research Council of Thailand
NTFP	Non-timber Forest Product
SURUMER	Sustainable Rubber Cultivation in the Mekong Region
TRY	Tai Rom Yen National Park

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Chapter 1 General introduction

1.1 Thesis overview

Expanding human land-use systems and decreasing natural areas pose great challenges for biodiversity conservation throughout the world. In Southeast Asia, forests and traditionally managed agroecosystems have been replaced by cash crop plantations such as rubber and oil palm cultivations (Wilcove et al. 2013). Apart from habitat loss, increasing conflicts with people further threaten conservation efforts as wildlife enters the farmland in search for food and water. The endangered Asian elephant (*Elephas maximus*) is frequently in the focus of such conflicts as these pachyderms pose great threats to cultivated crops and people's lives and livelihoods in transition zones of cultural and natural areas (Sukumar 1989). At the same time, these megaherbivores hold key functions in forest ecosystems and serve as flagship and umbrella species for conservation actions (Campos-Arceiz & Blake 2011; Douglas & Veríssimo 2013).

Despite the ecological and economic importance of wildlife in natural and agricultural ecosystems, very little is known on the use of farm-forest transition zones by wild mammal species such as elephants in rubber-dominated areas. Meanwhile, better knowledge on wildlife diversity and habitat use in human-dominated landscapes is essential to ensure successful conservation efforts and to improve conflict mitigation approaches as well as sustainable plantation management.

To address this gap of knowledge, we combined ecological and socio-economic sampling methods in this thesis. We conducted transect and camera surveys as well as interviews with local farmers to assess wild mammal diversity, habitat use as well as the kind and extent of damage by wildlife in and around a protected area in southern Thailand. Furthermore, with feeding and seed germination experiments we investigated the ecological importance of elephants for the dispersal of an exemplary tree species and to draw inferences for megafaunal-syndrome plant communities. The obtained data were then used to support the assessment of ecosystem services including multiple species and functional diversity levels in a cultural landscape in Southeast Asia and to discuss options for a more sustainable and wildlife-friendly plantation management of rubber cultivation.

1. 2 Nature conservation versus expanding human land-use

1.2.1 The state of biodiversity in Southeast Asia

The wellbeing of humankind strongly depends on the biological diversity of our planet: biodiversity facilitates the provisioning of economic, social and esthetic benefits, be it in form of food, energy, medicinal plants or for recreation (WCMC 1992; MEA 2005). Paradoxically, the increasing human wellbeing and population growth in recent times has led to a severe and ongoing depletion of our world's biological resources and, therefore, the foundation of our lives (Vihervaara et al. 2010). Since the beginning of the Anthropocene, biodiversity has seriously declined with accelerated extinction rates possibly up to 1000 times higher than the background rate (Barnosky et al. 2011; Pimm et al. 2014; Williams et al. 2015). Main drivers of declines in biodiversity are the loss, fragmentation and degradation of natural forest, environmental pollution, the overexploitation of natural resources, invasive species and climate change (CBD 2010).

The greatest loss of forest since 1990 occurred in the tropics, where the forests harbor the highest terrestrial levels of biological diversity (WCMC 1992; Martin & Sauerborn 2013; FAO 2015). Between 1990 and 2010, an estimated 121 million ha of tropical forest were lost – an area more than three times the size of Germany (Achard et al. 2014). In Southeast Asia, which comprises several biodiversity hotspots, not even half of the original forests persisted until nowadays while the degradation of the remaining natural habitats is ongoing (Myers et al. 2000; Sodhi et al. 2004; Hansen et al. 2013; FAO 2015). More than 40% of the region's biodiversity could be lost by the end of this century (Sodhi et al. 2004). As large mammals are often the first animals to disappear due to hunting and habitat disturbance, even up to 48% of mammalian species in Southeast Asia could go extinct until 2100 (Brook et al. 2003; Corlett 2007; Markl et al. 2012; Ripple et al. 2016a; Ripple et al. 2016b). If we continue to lose further species at the current rate, a sixth mass extinction will be inevitable, which means that more than 75% of species would vanish within a relatively short time period of not more than 2 million years (Myers et al. 2000; Brook et al. 2003; Sodhi et al. 2004; Magurran & Dornelas 2010; Barnosky et al. 2011; Pimm et al. 2014; Cafaro 2015). In the case of mammals, such a mass extinction could already be reached by around 300 to 1700 years from now with far-reaching negative ecological and socio-economic consequences (Barnosky et al. 2011).

Wild mammal diversity is an important component of overall biodiversity which is essential for the functioning of ecosystems and the provisioning of services derived from nature (MEA 2005; Turner & Daily 2008; Malhi et al. 2016). Exemplary ecosystem functions (ESF) comprise primary production, nutrient storage or hydrological cycles and derived ecosystem services (ESS) include the provisioning of food, medicinal products or water supply and regulation (Costanza et al. 1997; MEA 2005; TEEB 2012). Mammalian herbivores and predators function as primary and secondary regulators in ecosystems and the services they provide include habitat provision, food, pollination, biological control and recreation (TEEB 2012; Malhi et al. 2016). Large vertebrates further contribute significantly to nutrient fluxes and to the maintenance of carbon storage in tropical forests due to their services as seed dispersers (Wolf et al. 2013; Bello et al. 2015). Quantifying such services in economic terms is difficult but the average value of merely 17 ESS is estimated to be a minimum of 33 trillion US$ per year while the global gross national product is around 18 trillion US$ per year (Costanza et al. 1997).

The best option to protect biodiversity as well as associated ecosystem functions and services is through conserving natural areas and currently a total of 202,467 protected areas cover 14.7% of the global land and inland water surface (Western & Waithaka 2005; UNEP-WCMC and IUCN 2016). Still, the extent and connectivity of these protected sites is not sufficient to cover the distribution of all threatened species with about 20% being left unprotected concerning their habitat (Rodrigues et al. 2004). Successful protection of biodiversity, therefore, also increasingly depends on the conservation efforts in farmland surrounding protected areas (Daily 2001; Harvey et al. 2008; Laurance et al. 2012; Baudron & Giller 2014).

1.2.2 Rubber cultivation and its impacts

Over the last decades the area under rubber cultivation has expanded massively, particularly in Southeast Asia, with currently more than 10 million ha covered by this cash crop (Li & Fox 2012; FAOSTAT 2016). The rubber plant *Hevea brasiliensis* is a fast-growing tropical tree of the family Euphorbiaceae, which originates from the Amazon basin in South America from where it was exported to Southeast Asia at the end of the 19th century (Priyadarshan 2011). Starting from an age of about 5 to 8 years, the trees are tapped for their latex, a milky dispersion of rubber particles, which runs through vessels in

the bark (Jones & Allen 1992; Premakumari & Panikkar 1992). The economic lifespan of rubber plantations lies between 25 to 35 years after which the trees are cut and replaced (Priyadarshan 2011). While there are several other rubber-containing plants apart from *H.brasiliensis*, the latter is by far the most economically significant source of natural rubber (Jones & Allen 1992). Although the majority of the worldwide rubber demand is met by synthetic rubber, natural rubber with its unique properties remains an essential commodity, particularly in the car industry and mainly for the production of tires (Knoke et al. 2015; Jumpasut 2016a).

The increasing demand for natural rubber fuelled a rapid production growth and within the last 25 years the annual global production quantity more than doubled to around 12 million tons nowadays (FAOSTAT 2016; Jumpasut 2016b). More than 90% of rubber is produced in Asia while the rubber industry in South America is severely hampered by the South American leaf blight (SALB), a disease caused by the fungus *Pseudocercospora ulei*, which so far is absent from Asia (Liyanage & Kuruvilla 1992; Hora Júnior et al. 2014; Golbon et al. 2015; FAOSTAT 2016). Thailand and Indonesia are by far the leading producer countries, together accounting for almost 60% of the world production in natural rubber (FAOSTAT 2016).

Figure 1.1. Rubber-dominated landscape in Southwest China: Banana plantations in the valley and rubber plantations encroaching into natural forest on the hillsides.

The rubber boom in recent years has had multiple economic, social and ecological impacts in Southeast Asia. Rubber became an important source of income for many smallholder farmers and contributed to poverty alleviation in several areas (Liu et al. 2006; Manivong & Cramb 2008; Häuser et al. 2015). However, fluctuations in the market price leave rubber growers highly vulnerable as they often rely on this single crop for their income for many years to come (Häuser et al. 2015; Knoke et al. 2015). The decreasing crop diversity as a result of the change from traditionally managed agroecosystems to monoculture rubber plantations in combination with diminishing natural food resources might also have negative implications for rural food and livelihood security (Xue et al. 2005; Fu et al. 2010).

The ecological implications of the extensive rubber cultivation are manifold, leading to changes in the regional climate, the hydrological cycles and carbon stocks (Häuser et al. 2015). These alterations include increases in temperatures as well as a decrease in fog levels and reduced soil water contents linked to high evaporation rates (Wu et al. 2001; He & Zhang 2005; Mann 2009; Qiu 2009; Guardiola-Claramonte et al. 2010; Tan et al. 2011). The use of fertilizers and pesticides further deteriorated water quality and enhanced soil loss and erosion rates (Wu et al. 2001; Häuser et al. 2015; Liu et al. 2016). Depending on factors such as the topography, age and plantation management, carbon gains generally result from the conversion of intensively managed crop areas to rubber farms while losses occur when natural forest is transformed into rubber plantations (Ziegler et al. 2012; Blagodatsky et al. 2016).

The transformation of natural forest to rubber plantations unequivocally results in decreased overall levels of biodiversity (Aratrakorn et al. 2006; Behm et al. 2013; Li et al. 2013; He & Martin 2015). The diversity of insectivorous bats, for example, was reduced by two thirds in rubber plantations in Thailand compared to natural forest and 22 times more individuals were captured in the latter habitat (Phommexay et al. 2011). In contrast, small non-volant mammal diversity in general seemed to be not much affected by rubber plantations in Malaysia (Nakagawa et al. 2006). Few studies are available for larger mammals in rubber plantations but lower species numbers of primates were found in the cultivated areas compared to natural habitat in Sumatra while more than 70% of larger mammal species that occur in natural forest were still found in plantations in Sri Lanka (Gamage et al. 2011).

The degradation of biodiversity in rubber-dominated landscapes does not only affect plants and animals but is also of concern for humans who benefit from goods and services provided by intact ecosystems (Costanza et al. 1997; Balmford et al. 2002; Isbell et al. 2011). While rubber plantations cannot fulfill the same ecosystem functions or services as natural forests, management of cultivated land nevertheless needs to be improved to sustain the provisioning of essential services which depend on the conservation of biodiversity.

1.2.3 The SURUMER project

The project 'SURUMER' stands for 'Sustainable Rubber Cultivation in the Mekong Region' and its overall aim is the development of an integrative, applicable and stakeholder-validated land-use concept for sustainable rubber cultivation in the Yunnan Province in China, which can be transferred to other areas of the Greater Mekong Region (https://surumer.uni-hohenheim.de). The research presented in this dissertation is closely linked and partly embedded into the framework of SURUMER as subproject (SP) number 6 concerning 'Nature Reserve Conflicts'. A total of nine SPs comprising German, Chinese and Thai researchers from different disciplines have worked together to define multi- and interdisciplinary concepts for the integration of ESF/EES conservation efforts with socio-economic targets and drivers. Some of the major challenges addressed by SURUMER concern the altered ESF/EES in the hydrological cycle, in carbon dynamics or in soil fertility levels as well as threats to natural and agro-biodiversity. Furthermore, the project assesses risks and chances for rubber farmers' livelihoods and options to consolidate short-term socio-economic needs with environmentally sustainable farm management options.

SURUMER started in December 2011 and will end by mid-2017. Most of the overall project's field work took place in Yunnan Province in China, namely within the Naban River Watershed National Nature Reserve (NRWNNR). Our SP 6, however, expanded research activities to Thailand as another member country of the Greater Mekong Region and due to permit restrictions in China concerning research on larger wildlife in protected areas. We selected the Tai Rom Yen National Park (TRY) in Surat Thani Province in Southern Thailand as an exemplary conservation area within a traditional rubber-cultivating landscape, where resource competition with elephants had sparked increasing conflicts in recent years.

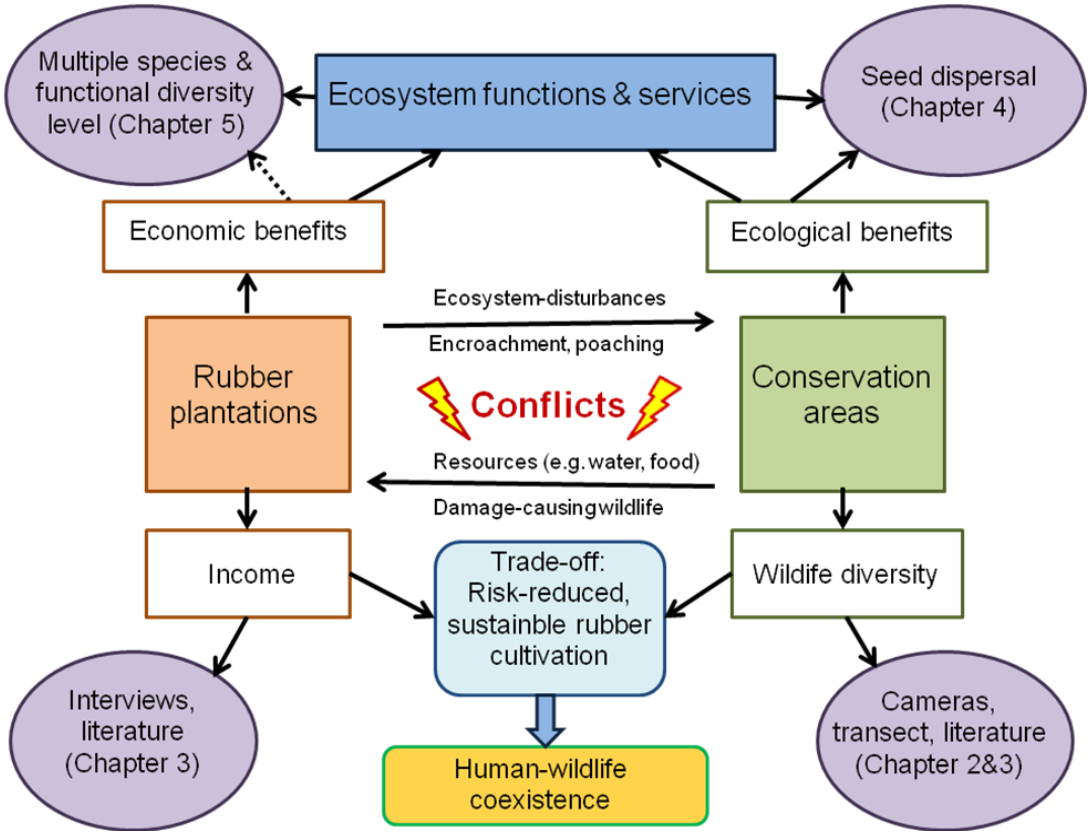

Figure 1.2. Functional framework of this thesis.

1.3 Conflicts with wildlife in transition zones of cultural and natural landscapes

Conflicts between people and wildlife occur as a result of overlapping needs and requirements with negative consequences for both sides (IUCN Worlds Park Congress (WPC) 2004; Distefano 2005). The fringes and transition zones of conservation areas are particularly prone to human-wildlife conflicts (HWC) as wildlife is dispersing from natural areas into adjacent farmland where human settlement and population growth is especially high (Boafo et al. 2004; Thirgood 2005; Oppong et al. 2008; Wittemyer et al. 2008; Wilson et al. 2015).

Each year, hundreds of people are killed by wildlife, constituting the most severe type of HWC (Thirgood et al. 2005). Amongst the species most often reported to be responsible for attacks are large felines such as tigers (*Panthera tigris*) and lions (*P. leo*) but also megaherbivores such as hippos (*Hippopotamus amphibious)* and elephants (*Loxodonta africana, L. cyclotis, Elephas maximus;* Nyhus & Tilson 2004; Distefano 2005; Quigley &

Herrero 2005; Dunham et al. 2010; Acharya et al. 2016). In Africa, crocodiles (*Crocodylus niloticus)* are likely responsible for more deaths than any other wildlife species (Lamarque et al. 2009; Dunham et al. 2010). Fatal encounters between people and wildlife are relatively rare compared to other causes of human death but are traumatic for affected families and communities and have strong negative effects on conservation efforts (Quigley & Herrero 2005; Thirgood et al. 2005).

The most prevalent forms of conflicts are livestock predation and crop foraging by wild animals (Thirgood et al. 2005; Lamarque et al. 2009). Distance from the natural area boundary is one of the most significant predictors for crop damage (Barnes et al. 1995; Naughton-Treves & Treves 2005; Sarker & Røskaft 2010; Wilson et al. 2015). For example in Africa, around the Kibale National Park in Uganda, 4 to 7% of different crops were lost to various wildlife species per season in 1992 to 1994 within half a km from the forest and 90% of this damage occurred only within 160 m from the forest edge (Naughton-Treves 1997). In Asia, around the Kerinci Seblat National Park in Sumatra, wild pigs (*Sus scrofa*) and pig-tailed macaques (*Macaca nemestrina*) damaged crops in an area of around 1400 m^2 within 5 months from 2001 to 2002 (Linkie et al. 2007).

Charismatic and large species like elephants are frequently in the focus of attention but small and abundant species such as rodents have a more significant overall economic impact concerning crop damage (Thirgood et al. 2005; Wood & Singleton 2015; Nyhus 2016). However, damage levels strongly depend on the geographical scale and while crop loss to larger wildlife species plays a negligible role on a national level, the extent of damage in local and individual situations can be devastating for affected farmers (Naughton-Treves 1998). Local people also feel more vulnerable to large animals because of the risk of being attacked and as such species are often protected under national and international law, thereby prohibiting farmers to take action against this 'government property' at their own measure (Naughton-Treves 1997; Thirgood et al. 2005; Lamarque et al. 2009; Douglas & Veríssimo 2013).

Some of the major causes leading to biodiversity loss are also fuelling conflicts with larger wildlife species. The growing demand for land and resources as a result of the increasing human populations leads to diminishing distances between the living space of people and the habitat of wildlife with associated risks for conflicts. This is particularly the case in

densely populated Asia where 60% of the world's population live but also in Africa where the population growth rate is higher than anywhere else with a projected increase of 1.3 billion people until 2050 (UN 2015).

Apart from decreasing and degrading natural habitat it is also the transformation of land uses within the agricultural landscape that can contribute to rising conflict levels. Wild pigs are a common example for increasing population numbers as a result of overabundant food supplies, e.g. in form of monoculture maize or oil palm plantations (Ickes 2001; Hua et al. 2016). The absence of predators to control prey populations further fosters such hyper-abundances of certain species (Ickes 2001). Unbalanced prey-predator ratios in form of insufficient prey species are common sources of conflicts as well as carnivores find easy alternatives in the form of livestock with its growing numbers (Khorozyan et al. 2015; Miller et al. 2016). Some wildlife populations have also seen increases as a result of successful conservation programs, which can lead to problems if the natural habitat locally available becomes too small (Messmer 2000; Kirkpatrick et al. 2012).

The consequences for people competing with wildlife for food and space are first of all safety risks for their lives and livelihoods. Apart from the danger of death and injuries, the risk of zoonotic diseases such as rabies or malaria increases through more frequent contact with wildlife (Wang et al. 2014; Divis et al. 2015). Crop depletion by wild animals can lead to severe food shortages with associated health impacts (Mackenzie & Ahabyona 2012; Harich et al. 2013). For example, the food security in communities around the Queen Elizabeth National Park in Uganda was reduced by 14% annually due to crop-raiding, mainly by elephants (Barirega et al. 2010). Property damage other than crops, e.g. to houses and vehicles, leads to further high economic burdens (Distefano 2005; Pant et al. 2016). The social and indirect costs are less obvious at first glance but can seriously affect the life quality of communities as the danger of encountering wildlife or time-consuming guarding duties can lead to restrictions in everyday life activities such as school attendance of children (Haule et al. 2002; Kagoro-Rugunda 2004; Ogra 2008; Mackenzie & Ahabyona 2012).

Consequences for wildlife crossing the path of human interests are equally dire. Individual animals get killed in retaliation or if they are classified as problem animals (Treves & Naughton-Treves 2005; Parker et al. 2007; Kissui 2008). Exacerbating conflicts can lead to

the eradication of local populations and can pose a serious threat to the survival of single species as well as overall biodiversity conservation (Woodroffe et al. 2005; Lamarque et al. 2009; Nyhus 2016). The latter might particularly become of significance if the species in the conflict focus fulfills a keystone role in natural systems and its decline or removal could lead to disturbances in an ecosystem's functioning as is the case with elephants (Jones et al. 1994; Pringle 2008; Campos-Arceiz & Blake 2011).

1.4 The case of Asian elephants – ecological and economic importance

Elephants are one of the most ambiguous wildlife species from a human perspective. Highly revered and loved by one part of the global population and deeply feared and hated by other groups, mainly those sharing the same resources with the pachyderms. Elephants are at the same time keystone, flagship and umbrella species but also a serious agricultural pest to rural farmers (Blake & Hedges 2004; Perera 2009; Barua 2011; Douglas & Veríssimo 2013).

There are three extant species of elephants, the African savannah (*L.africana*), the African forest *L. cyclotis*) and the Asian elephant (*E. maximus*). Population numbers of all three species are dwindling in the face of increased poaching for ivory, habitat loss and conflicts with people. A recent continent-wide survey estimated the population of African savannah elephants to be approximately 350,000 animals with a decline of 8% per year and an overall reduction of 144,000 animals between 2007 and 2014 (Chase et al. 2016). African forest elephants are more difficult to monitor but their population likely declined around 62% from 2002 to 2011, with around 100,000 individuals remaining (Maisels et al. 2013).
Asian elephants have the smallest population size amongst the three species, but the estimated number of 40,000-50,000 animals of the remaining population is hardly more than a rough guess (Sukumar 2003; Blake & Hedges 2004; Choudhury 2008). In any case, it is certain that Asian elephants are threatened by extinction (IUCN 2016). Based on current and historic distributions and habitat quality their population size has likely been reduced at least by 50% over the last three elephant generations (Choudhury 2008).

The geographic range of Asian elephants once stretched from the Yangtze River in China through Southeast Asia into India and as far as Mesopotamia (Sukumar 2003). Nowadays, the distribution is restricted to 13 countries with India holding the largest population

(Sukumar 2006). In almost every place where elephants occur, conflicts are on the rise (Perera 2009; Zhang 2011; Madhusudan et al. 2015; Pant et al. 2016).

The damage elephants can inflict on people, crops, livestock and property is tremendous. In China, where less than 250 wild elephants persist in the Southwest, more than 18,000 conflict incidents were recorded from 2008 to 2012 of which 19 were attacks on people (Zhang et al. 2015; Chen et al. 2016). And in Karnataka state in India with probably the largest elephant population in Asia, close to 70,000 crop-raiding incidences were recorded from 2008 to 2011 and 91 people died as a result of encounters with the pachyderms (Gubbi et al. 2014). At the same time, around 100 elephants are killed each year in India of which more than half are a result of crop defense by humans while other common causes for deaths in Asian elephants are poaching, diseases, collision with vehicles and electrocution (MOEF 2010; Lin et al. 2011; Palei et al. 2014; Davidar et al. 2015; Fig. 1.3).

In view of the overabundant damage incidents it is often forgotten that elephants do not only induce costs but can also contribute to economical benefits in form of tourist revenues. Estimates of economic benefits lost due to elephant poaching in African savannahs amount to 25 million USD per year (Naidoo et al. 2016). Although such revenues are not comparable to Asia where wildlife observations are more difficult due to the forest-type of habitats, wild elephants can nevertheless constitute a major touristic attraction in some areas (Tisdell & Zhu 1998; Buultjens et al. 2005; Zhang 2011; Aihara et al. 2016).

Figure 1.3. Fruit tree damaged by wild elephants and female elephant electrocuted by a fence erected by farmers to protect their crops (pictures taken in southern Thailand).

Apart from such economic benefits, elephants do further provide many ecological services to their environment. The megaherbivores consume and destroy huge amounts of plant material every day, thereby creating light and space for fresh plant re-growth which can benefit smaller wildlife species as well (Wegge et al. 2006; Pringle 2008; Bakker et al. 2016; Terborgh et al. 2016). They further play an important role as seed dispersers of various plant species and the continuous decline of elephant populations will likely lead to major shifts in forest structures and lower overall levels of biodiversity in ecosystems with implications for the services latter can provide to humankind (Guimarães Jr. et al. 2008; Blake et al. 2009; Campos-Arceiz & Blake 2011; Isbell et al. 2011; Coverdale et al. 2016).

1.5 Objectives

The main objective of this thesis was to analyze the effects of rubber cultivation on the species richness, presence and distribution of wild mammal populations and to investigate the potential of rubber-dominated landscapes to sustain wild mammal diversity and associated ecological functions in proximity to natural areas, with a focus on conflict mitigation. The specific objectives were to

(1) Identify the species richness of wild mammals in the transition zone of natural forests and rubber-dominated land-use systems

(2) Assess the presence and habitat use of mammalian wildlife in rubber-dominated landscapes

(3) Analyse existing and emerging conflicts between humans and wildlife as well as associated economic losses in an exemplary rubber-dominated landscape surrounding natural areas

(4) Determine the ecological importance of conserving large wildlife species using the example of Asian elephants and their seed dispersal potential

(5) Develop a methodology for incorporating multiple levels of species diversity into integrated ESS evaluation models

1.6 Outline of the thesis

This dissertation is a cumulative thesis, with each chapter representing a published or a submitted peer-reviewed journal article. Chapter 2 is a review giving an overview of the current knowledge concerning the diversity of wild mammals in rubber plantations and the potential of such landscapes to act as extended habitat for wildlife assemblages including risks of conflicts. As data on rubber cultivation were scarce, we included oil palm landscapes as another, usually large-scale monoculture type in the review as comparison from which general ecological impacts might be deduced.

Chapter 3 provides the outcomes of our own data collection in Thailand where we assessed the diversity and presence of wild mammal species in a traditional rubber-cultivating landscape in southern Thailand. We further analyzed the kind and extent of wildlife damage using transect and camera surveys as well as interviews with local farmers and discuss management implications.

In Chapter 4, we investigate the seed dispersal potential of Asian elephants using an exemplary megafaunal syndrome fruit species. Through feeding and germination experiments we aimed to determine the ecological importance of the megaherbivores to highlight the benefits of conserving large mammals for entire ecosystems.

Chapter 5 consolidates the multi-disciplinary work and outcomes of our SURUMER project for the design and development of an integrated ESS assessment model that incorporates different levels of biodiversity. The presented method provides opportunities for comparable impact assessments under different land-use scenarios and can be transferred to other regions and land-use types.

Chapter 6 discusses the main findings of this thesis and the implications and recommendations for future management options regarding sustainable rubber cultivation and wildlife conservation.

Chapter 2

Mammalian wildlife diversity in rubber and oil palm plantations[1]

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2.1 Abstract

In face of globally diminishing natural habitats in biodiversity-rich regions, agricultural landscapes around protected areas have increasingly gained importance as extended habitat for wildlife species. Rubber (*Hevea brasiliensis*) and oil palm (*Elais guineensis*) plantations are two of the dominant land-use systems in Southeast Asia that have seen a tremendous expansion over the last decades. Despite far-reaching ecological consequences of these intensively cropped monocultures on natural ecosystems, relatively little is known about their utilization by wildlife populations. With this review we want to give an overview of mammalian diversity in rubber and oil palm plantations with reference to human-wildlife conflicts occurring as a result of overlapping resource use. We searched the literature for studies on wild mammalian diversity in rubber and oil palm plantations and found 17 publications. We considered 29 additional publications that provided information on single species in such plantations. We discuss the potential of 'wildlife-friendly' farming for mammalian assemblages in plantations and its importance in the case of rubber and oil palm production. Our review showed that most wild mammal species found in these plantations were likely to be visitors that use cultivated landscapes as fringe habitat but some adapted well to plantations and few even became resident. We conclude that although plantations in the tropics and subtropics cannot substitute for forests and the preservation of natural habitats is indispensable, the reality of ongoing forest degradation and transformation into plantations will make wildlife-friendly farming a key strategy in

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maintaining mammalian diversity, particularly in land-use matrices surrounding natural habitats.

Keywords: *Rubber, oil palm, wildlife, biodiversity, wildlife-friendly farming, human-wildlife conflicts, land use*

2.2 Review Methodology

We searched the Scopus database and the Web of Science for the basic keywords 'rubber', 'oil palm', 'biodiversity','wildlife' and 'wildlife-friendly' in combination with each other. Furthermore, we used relevant references from the literature reviewed and searched for relevant material on the social networking site 'ResearchGate' as well as on Google scholar. We also contacted eleven experts and authors to ask for more information and material on the topic. These experts were chosen based on their publications which indicated further unpublished data were available or according to their expertise in wildlife research known through personal connections with the authors of this review. We did not limit the review to a specific region but as most natural rubber and palm oil is produced in Asia and particularly Southeast Asia, our review has a strong focus on this region.

2.3 Introduction

The extent to which tropical and subtropical forests have been degraded and transformed into agricultural plantations over the last decades is unprecedented and poses serious threats to large parts of our planet's fauna and flora (Laurance 1999; Achard et al. 2002; Sodhi et al. 2004). From 1990 to 2015, the net loss of global forests amounted to 129 million ha and the decline is still ongoing (FAO 2015). With the reduction in natural forest area, fewer habitats are available for wildlife. Particularly wild mammal species are very sensitive to natural habitat disturbances (Sodhi et al. 2009). In Southeast Asia, large areas of forest as well as traditionally managed agroecosystems have been cleared and transformed into monocropped cash crop cultivations, particularly into rubber and oil palm plantations (Wilcove et al. 2013). Nowadays, less than half of the original forests are remaining in Southeast Asia (Sodhi et al. 2004) while more than ten million ha of land are covered by rubber trees and more than 18 million ha land are cultivated with oil palm (Rivano et al. 2013; FAOSTAT 2015a). As a result, it has been estimated that about 21 to

48% of mammals in Southeast Asia might go extinct until the end of this century (Brook et al. 2003). Southeast Asia produces around 75% of all rubber and 86% of all palm oil currently on the market (FAOSTAT 2015b). Natural rubber is an essential product for the vehicle industry, especially for high stress purposes as in the case of airplane and truck tires (Mann 2009). Oil palm, apart from being an important commodity in the food industry, has seen an additional boost as source of biofuel in recent years (Yaap et al. 2010). These economic benefits were obtained at the expenses of the region's rich biodiversity which suffered dramatic losses (Sodhi et al. 2004; Danielsen et al. 2009; Yaap et al. 2010). Species diversity in oil palm plantations was found to be reduced by 85% compared to primary forest (Fitzherbert et al. 2008; Fargione et al. 2010).

Both crops thrive best in areas that are naturally covered with tropical rainforest and that belong to the most important biodiversity hotspots (Myers et al. 2000). Increasing prices for palm oil has led to an expansion of the crop into traditional rubber cultivation areas as palm oil production is confined to low and humid areas (Corley & Tinker 2003; Fox & Castella 2013), while the development of hybrid rubber trees nowadays allows rubber production in higher elevations and latitudes, leading again to an expansion of rubber into new, often naturally vegetated or traditionally managed areas (Li & Fox 2012). Although more than two thirds of rubber plantations in continental Southeast Asia are currently placed in marginal zones where lower yields are to be expected (Ahrends et al. 2015), simulations nevertheless forecast an increase in land covered by rubber of about 2.5% (42,500 km^2) in montane mainland Southeast Asia until 2050, thereby further replacing forests and more diverse swidden cultivation systems (Fox et al. 2012). However, it remains to be seen how the recent decrease in rubber prices will affect this development.

This looming biodiversity disaster prompted calls for urgent counteractions (Sodhi et al. 2004), naturally with accompanying disagreement over the appropriate measures as is to be expected for the dimensions of such multiple faced challenges. Two contrasting concepts for biodiversity conservation are represented by 'land sparing' and 'land sharing' (also termed wildlife-friendly farming) approaches (Green et al. 2005; Fischer et al. 2008; Baudron & Giller 2014). Land sparing refers to the idea of the exclusion of intact habitats from use while increasing the production on the actual farmland, thereby avoiding the need for further conversion of natural areas. In contrast, land sharing aims at facilitating an

increased biodiversity within farmland through the provision of natural elements, an overall higher heterogeneity or a lower input of fertilizer and pesticides.

A driving motivation behind the land sharing idea is the challenge that global protected area networks are not sufficient to conserve all species, as all protected areas combined only cover 15.4 % of the terrestrial surface (Juffe-Bignoli et al. 2014) and include only potential habitat for around 80% of all threatened species (Rodrigues et al. 2004). With decreasing natural habitat many species have no choice but to partially adapt to plantations as extended habitat in order to survive. Their success depends on the structure and kind of farmland, especially in areas surrounding protected areas and natural forests (Chazdon et al. 2009). Specialist species are struggling to adapt and further declines are likely while generalist species might even benefit from modified landscapes (Yaap et al. 2010). Conversion of primary forest to plantations generally results in decreased diversity of wildlife communities in the tropics and subtropics (Danielsen & Heegaard 1995; Gamage et al. 2011; Freudmann et al. 2015). In Europe, the agricultural activities themselves had led to an increase in biodiversity in the first place due to their increasing habitat heterogeneity, which had replaced the formerly more homogenous forested landscapes. However, only recently agriculture became one of biodiversity´s biggest threats in Europe (Martin & Sauerborn 2013). On the other hand, newly established cultivations in the tropics and subtropics cannot by any means substitute for the high biodiversity found in natural tropical forest (Martin & Sauerborn 2013).

With this review we give an overview of the current knowledge concerning the potential of rubber and oil palm dominated landscapes as extended habitats for mammalian wildlife assemblages, mainly in Asia where most available studies were conducted. We focus on mammals only as we feel that there is a gap of knowledge for this taxon, particularly in rubber-dominated landscapes, while impacts of cultivations on general biodiversity were addressed before (e.g. Fitzherbert et al. 2008; Danielsen et al. 2009; Yaap et al. 2010). For a collection of species records in rubber and oil palm plantations we considered data from 46 publications, 39 of which were peer-reviewed (Olivier 1978; Blair & Noor 1981; Wood & Liau 1984; Sukumar 1989; Danielsen & Heegaard 1995; Buckle et al. 1997; Rajaratnam 1999; Medhi et al. 2004; Normua et al. 2004; Nakagawa et al. 2006; Dela 2007; Maddox et al. 2007; Nasi et al. 2008; Bernard et al. 2009; Fukuda et al. 2009; Marchal & Hill 2009; Campbell-Smith et al. 2011a; Campbell-Smith et al. 2011b; Dela 2011; Gamage et al.

2011; Phommexay et al. 2011; Puan et al. 2011; Shafie et al. 2011; Weerakoon & Wickramasinghe 2011; Zhang 2011; Campbell-Smith et al. 2012; Dela 2012; Estes et al. 2012; Sunarto et al. 2012; Azhar et al. 2013a; Chen et al. 2013; Nakashima et al. 2013; Wahyudi & Stuebing 2013; Azhar et al. 2014a; Bernard et al. 2014; Edwards et al. 2014; Luskin et al. 2014; Ancrenaz et al. 2015; Azhar et al. 2015a; Freudmann et al. 2015; Jennings et al. 2015; Morand et al. 2015; Wood & Singleton 2015; Yue et al. 2015). The contacts to experts in the field of wildlife research yielded two additional publications previously not found in the literature search (Normua et al. 2004; Wahyudi & Stuebing 2013), and four unpublished species lists, three of which were referred to in the published articles (Gamage et al. 2011; Weerakoon & Wickramasinghe 2011; Wahyudi & Stuebing 2013; Morand et al. 2015). Only 17 studies had a focus on mammal biodiversity in rubber or oil palm plantation, 15 of which provided comparisons with more natural habitats, while the other 29 publications focused on other topics such as human-wildlife conflicts or on the ecology of particular species (Table 2.1). For the 15 studies, we graphically highlighted which species had been observed in rubber versus oil palm plantations and compared these numbers to wildlife numbers in adjacent natural habitat (see Fig. 2.1). We discuss different options for wildlife-friendly farming in oil palm and rubber cultivations and suggest suitable measures for the two cash crops.

2.4 Mammalian wildlife diversity in rubber and oil palm plantations

The impacts of habitat degradation on bird communities in Southeast Asia are comparably well documented in contrast to the effects on mammalian assemblages (Sodhi & Brook 2006). Several studies investigated the bird species diversity in rubber and oil palm plantations (Yorke 1984; Aratrakorn et al. 2006; Peh et al. 2006; Beukema et al. 2007; Edwards et al. 2010; Azhar et al. 2011; Azhar et al. 2013b; Li et al. 2013; Azhar et al. 2014b; Teuscher et al. 2015) but very little is known to what extent mammalian wildlife species are using such landscapes (Chazdon et al. 2009; Park 2015), despite their importance in ecosystems as herbivores, seed dispersers or predators. Evidence of wild animals on farmland are often reported as a result of conflicts in form of attacks to humans or livestock or more common in the case of crop loss (Nyhus & Tilson 2004a; Feng & Zhang 2005; Dela 2011; Campbell-Smith et al. 2012; Chen et al. 2013). Mammalian species assessments in plantations per se are scarce. To our knowledge, only five studies focused on mammal assemblages in rubber plantations (Danielsen & Heegaard 1995;

Nakagawa et al. 2006; Gamage et al. 2011; Phommexay et al. 2011; Weerakoon & Wickramasinghe 2011) while we were able to retrieve 17 assessments of wildlife biodiversity in oil palm plantations (Danielsen & Heegaard 1995; Rajaratnam 1999; Maddox et al. 2007; Bernard et al. 2009; Fukuda 2009; Gamage et al. 2011; Shafie et al. 2011; Weerakoon & Wickramasinghe 2011; Azhar et al. 2013a; Wahyudi & Stuebing 2013; Azhar et al. 2014a; Bernard et al. 2014; Edwards et al. 2014a; Azhar et al. 2015a; Freudmann et al. 2015; Jennings et al. 2015; Yue et al. 2015). Several other papers provide evidence of wildlife found in rubber or oil palm landscapes while their focus lies elsewhere (e.g. Olivier 1978; Marchal & Hill 2009; Campbell-Smith et al. 2011a; Campbell-Smith et al. 2011b; Dela 2011; Chen et al. 2013; Goswami et al. 2014; Table 2.1).

Without consideration of frequency or abundance, 109 mammal species records have been reported from these two cash crop plantations (see Table A.1 in the appendix for species list). Fifty-seven species were recorded in rubber, 23% of which were threatened according to the IUCN red list and 80 were reported from oil palm plantations, including 28% of threatened species, probably reflecting the higher number of studies available for the latter plantation type. However, many records represent occasional evidence. In Sumatra, Indonesia, 45% of species occurring in the region were found within oil palm plantations but only 10% were recorded to be regular visitors within the plantations (Maddox et al. 2007). Naturally, the biodiversity in farmland depends on local circumstances such as the surrounding landscape matrix and the plantation management itself. Distance from natural forests is often a predictor for decreasing biodiversity on farmland as wildlife found in monocultures is often constrained to the farm-forest interface (Bennett 2003; Maddox et al. 2007; Gubbi 2012; Gilroy et al. 2014; Yue et al. 2015). Rubber agro-forest or ‘jungle rubber’, in contrast, was found to support almost all forest species (Michon & de Foresta 1995). In the case of monocultures, most species are likely to be visitors only that use cultivated landscapes as extension of their core habitat, which is especially true for larger and threatened species (Azhar et al. 2013a).

Nevertheless, some species are able to cope well with human-modified landscapes. An isolated but increasing population of Golden langurs (*Trachypithecus geei*) occupied a rubber plantation for more than 17 years in Assam in India (Medhi et al. 2004) and rubber seeds, leaves, flowers and bark constituted an important food source for primates such as orangutans (*Pongo abelii*) in Sumatra (Campbell-Smith et al. 2011b) or purple-faced leaf

Table 2.1. Studies related to mammalian wildlife in rubber or oil palm plantations that were considered for this review (N = 46; and % of all 46 in parentheses). In the first line (diversity assessment), all studies that provided an assessment of wild mammal diversity in rubber or oil palm cultivations are mentioned. The second line (conflict focus) includes all studies that provided information on species found in the cultivations but had a focus on conflict issues including hunting. In the third line (focus on particular species) those studies are mentioned that focused on a particular species but contained information of the species' use of cultivations. Some studies addressed more than one aspect. Study (Mohd. Azlan 2006) is not included in this table as its focus was on mammal diversity not within oil palm plantations but within a secondary forest surrounded by an oil palm scheme.

	Rubber	**Oil palm**	**References**
Diversity assessment	5 (11%)	17 (37%)	Danielsen & Heegaard 1995; Rajaratnam 1999; Nakagawa et al. 2006; Maddox et al. 2007; Bernard et al. 2009; Fukuda et al. 2009; Gamage et al. 2011; Phommexay et al. 2011; Shafie et al. 2011; Weerakoon & Wickramasinghe 2011; Azhar et al. 2013a; Wahyudi & Stuebing 2013; Azhar et al. 2014a; Bernard et al. 2014a; Edwards et al. 2014a; Azhar et al. 2015a; Freudmann et al. 2015; Jennings et al. 2015; Yue et al. 2015
Conflict focus	7 (15%)	10 (22%)	Blair & Noor 1981; Wood & Liau 1984; Buckle et al. 1997; Marchal & Hill 2009; Campbell-Smith et al. 2011a; Campbell-Smith et al. 2011b; Dela 2011; Puan et al. 2011; Campbell-Smith et al. 2012; Azhar et al. 2013a; Chen et al. 2013; Luskin et al. 2014; Wood & Singleton 2015;
Focus on particular species	11 (24%)	11 (24%)	Olivier 1978; Wood & Liau 1984; Sukumar 1989; Buckle et al. 1997; Mohd. Azlan et al. 2003; Medhi et al. 2004; Normua et al. 2004; Dela 2007; Campbell-Smith et al. 2011a; Campbell-Smith et al. 2011b; Dela 2011; Zhang 2011; Campbell-Smith et al. 2012; Dela 2012; Sunarto et al. 2012; Chen et al. 2013; Nakashima et al. 2013; Ancrenaz et al. 2015;

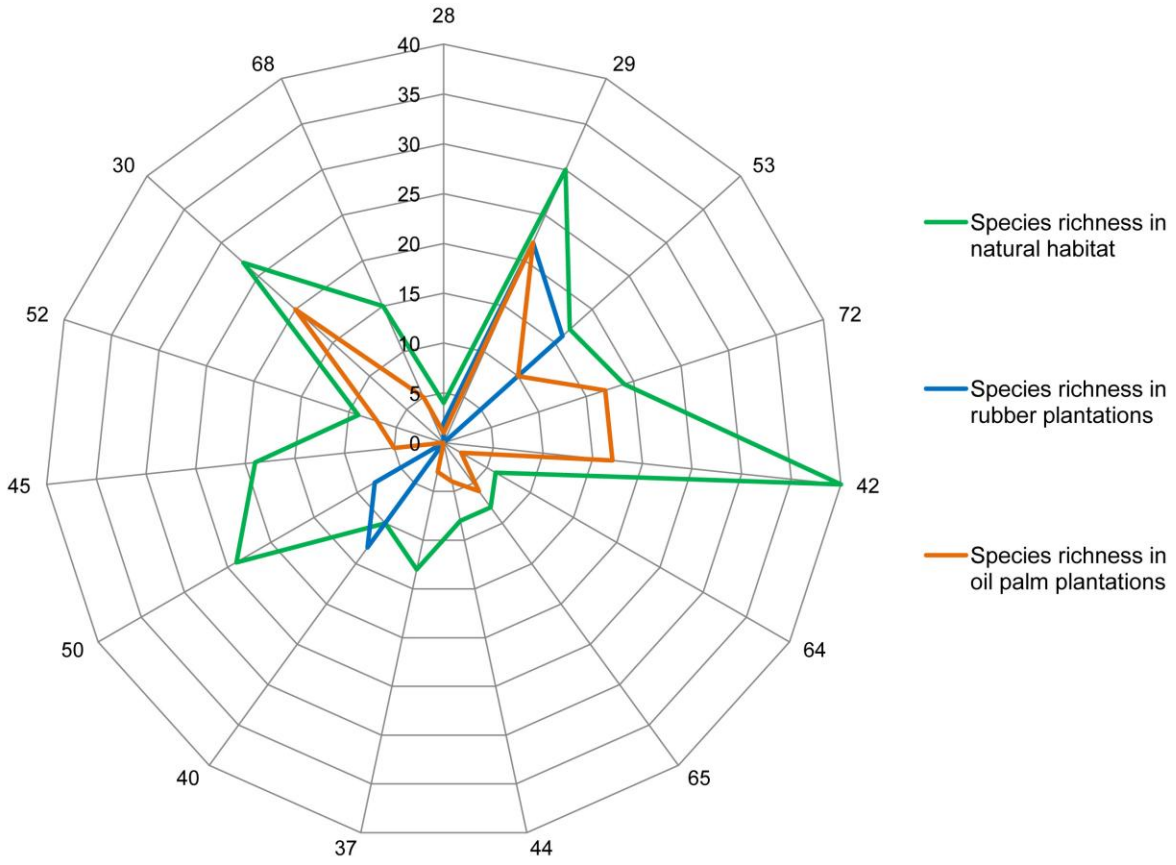

Figure 2.1. Number of mammal species found in natural habitat, in rubber or in oil palm plantations as mentioned by 15 studies. The outer numbers are references to the respective study while the vertical number indicates the number of species mentioned in each study. The colored lines represent mammal numbers found in natural forest habitat, rubber or oil palm plantations, respectively (with 0 standing for not considered in the respective study). For interpretation of the colored lines, the reader is referred to the electronic version of this article. Study 28 (Danielsen and Heegaard 1995) assessed primates, squirrels, treeshrews and bats, studies 29 (Gamage et al. 2011), 53 (Weerakoon and Wickramasinghe 2011), 72 (Yue et al. 2015) and 42 (Maddox et al. 2007) assessed all mammals or medium to large mammals, study 64 (Bernard et al. 2014a) looked at non-human primates, study 65 (Edwards et al. 2014a) assessed scavenging mammals, studies 44 (Bernard et al. 2009), 37 (Rajaratnam 1999), 40 (Nakagawa et al. 2006) researched non-volant small mammals and studies 50 (Phommexay et al. 2011), 45 (Fukuda et al. 2009), 52 (Shafie et al. 2011), 30 (Freudmann et al. 2015) and 68 (Azhar et al. 2015a) assessed bats, with no. 50 considering exclusively insectivorous bats, no. 30 phyllostomid bats and no. 68 exclusively fruit bats.

monkeys (*Trachypithecus vetulus*) in Sri Lanka (Dela 2012). Colobines (*Colobus* spp.) were found to occur in equal group numbers within a rubber plantation and in the natural forest in Sumatra; however, they were not recorded in oil palm (Danielsen & Heegaard 1995). Ungulates such as muntjacs (*Muntiacus* spp.) and sambar (*Rusa unicolor*) could benefit from a more open vegetation structure and higher herbaceous cover, especially while the planted crops are young and more understory vegetation is allowed to grow (McShea et al. 2009). In mature monocultures, less heterogeneity means reduced cover for ungulates, leaving them more susceptible to predation and hunting by carnivorous animals as well as by humans (Maddox et al. 2007).

Small mammal communities seem not to be affected severely by rubber plantations, at least in plantations where some natural vegetation is allowed to regenerate (Nakagawa et al. 2006). Rubber fruits can even serve as attractive food supply for small mammals (Nakagawa et al. 2006) as do oil palm fruits. Other species apart from rodents that feed on oil palm fruits or different parts of the plant might be resident or abundant in oil palm plantations, such as wild pigs (*Sus scrofa* and *S. barbatus*), macaques (*Macaca fascicularis* and *M. nemestrina*), porcupines (*Hystrix brachyura*) and squirrels (e.g. *Callosciurus* spp.) (Maddox et al. 2007; McShea et al. 2009; Azhar et al. 2013a). Leopard cats (*Prionailurus bengalensis*) benefit from the abundance of prey such as rats, amphibians, and reptiles (Maddox et al. 2007; Rajaratnam et al. 2007) and, therefore, might choose oil palm plantations as a major habitat type (Rajaratnam et al. 2007). Some civet species such as the common palm civet (*Paradoxurus hermaphrodites*) or the small Indian civet (*Viverricula indica*) can adapt well to the plantations due to their mixed diet that includes fruits, insects, amphibians and rodents (Maddox et al. 2007; Gamage et al. 2011). Also pangolins such as the critically endangered Sunda pangolin (*Manis javanica*) that feed on ants and termites dwell in oil palm plantations albeit they are severely hunted due to their high value in the illegal wildlife trade (Azhar et al. 2013a).

Like for other mammalian groups, diversity of bats is generally much reduced in rubber (Phommexay et al. 2011) and oil palm plantations compared to more natural habitats, while some few species can occur in high abundance in these monocultures (Fukuda et al. 2009; Azhar et al. 2015a; Freudmann et al. 2015). Depending on their morphological and ecological characteristics, frugivorous bat species differ in their ability to respond to human influences in the landscape (Saldaña-Vázquez et al. 2010). Some species prefer to

forage under low crop density (Azhar et al. 2015a) while others are found more abundant in denser crop stands (Estrada & Coates-Estrada 2002). Species richness of frugivorous bats benefits from a higher variety of plants in systems for food supply and shelter (Azhar et al. 2015a). The lesser short-nosed fruit bat (*Cynopterus brachyotis*) in particular dominates bat communities in many plantations (Danielsen et al. 2009; Fukuda et al. 2009; Shafie et al. 2011; Azhar et al. 2015a) and may even feed on oil palm fruits (Fukuda et al. 2009). The low abundance and diversity of insectivorous bats in oil palm and rubber stands is probably due to a lower abundance of insects (Fukuda et al. 2009; Shafie et al. 2011).

2.5 General threats of plantation farming to wildlife biodiversity

Apart from transforming natural habitat into simplified landscapes, plantations hold further threats to wildlife. Firstly, the structure and management of a plantation determines the possibility for wildlife to persist. Reduced landscape heterogeneity results in less diverse food availability for wildlife and reduced options for shelter. Increasing simplification through the homogenization of the environment and through the reduction in the structural and functional heterogeneity therefore leads to decreasing biodiversity (Danielsen & Heegaard 1995; Gamage et al. 2011; Martin & Sauerborn 2013). Also, the application of fertilizer, particularly in form of nitrogen, results in reduced plant species diversity, as does the use of herbicides (Pratt Jr 1984; Tilman 1987; Goldberg & Miller 1990; Mcclean et al. 2011; Martin & Sauerborn 2013). A reduced variety and number of plants and decreased ground cover negatively affects insect biodiversity (Meng et al. 2012a; Meng et al. 2012b) and, therefore, larger insectivorous animals. Impacts of fertilizer run-off on biodiversity are not restricted to terrestrial ecosystems but also impact the aquatic environment (Howarth et al. 2002; Galloway et al. 2003; Diaz & Rosenberg 2008; Gharibreza et al. 2013).

Where needs and demands of people and wildlife overlap, conflicts are likely to occur (IUCN Worlds Park Congress (WPC) 2004; Distefano 2005; Harich et al. 2013). Some species thrive so well in oil palm cultivations that they are considered a pest by plantation managers (McShea et al. 2009). The Malaysian field rat (*Rattus tiomanicus)* in particular is considered a major nuisance in oil palm plantations in Malaysia (Wood & Liau 1984; Buckle et al. 1997) which can cause estimated losses of about 5-10% of the average oil yield (Wood & Singleton 2015). Malayan black rats (*R. rattus diardii*) and the rice-field rat

(*R. argentiventer*) are two other abundant damage causing species, the latter one especially in younger plantations of up to five years old (Puan et al. 2011; Wood & Singleton 2015). Wild pigs (*S. scrofa*) profit greatly from the high energy oil palm fruits and increase their population densities accordingly (Ickes 2001), leading to damage that can exceed 500 USD/ha in some areas (Luskin et al. 2014). Hunting is, therefore, considered a standard control for this species, as well as for Malayan porcupines (*H. brachyura*) and macaques (*M. fascicularis* and *M. nemestrina*; Azhar et al. 2013a; Luskin et al. 2014). Apart from macaques, orangutans (*P. abelii*) are another damage causing species in oil palm plantations (Yuwono et al. 2007). In rubber, Thomas' leaf monkeys (*Presbytis thomasi*) and orangutans are frequent crop-raiders of rubber leaves (Marchal & Hill 2009) and rubber bark (Campbell-Smith et al. 2010; Campbell-Smith et al. 2011b), respectively.

The role of elephants as crop-raiders varies between different regions as they are not as abundant but locally more fatal once they are present due to their large size and their high food intake (Blair & Noor 1981; Chong & Dayang Norwana 2005; Alfred et al. 2012; Estes et al. 2012; Luskin et al. 2014). These pachyderms prefer young palms, which are then completely destroyed, while damage in older plants is less fatal (Blair & Noor 1981; Sukumar 1989; Chong & Dayang Norwana 2005). In South Pahang in Malaysia, elephants destroyed around 3000 acres of oil palm in 1980 (Blair & Noor 1981) and accounted for the third most costly reason of damage in the Jambi province in Sumatra in 2011 (Luskin et al. 2014). Damage of rubber by elephants affects mainly seedlings and saplings which are sometimes eaten but mostly represent collateral damage when elephants cross plantations (Chen et al. 2013). In an effort to protect crops, farmers often set up electric fences with uncontrolled electrical flow, in which elephants frequently die (Palei et al. 2014).

Sometimes responses to crop-raiding wildlife are retaliation killings (Nyhus & Tilson 2004b; Bhattarai & Fischer 2014). Hunting in general is a permanent threat to wildlife venturing into farmland. Farmers in Sumatra sometimes hunt mouse deer (*Tragulus* spp.) in rubber plantations; deer species in general are the second most harvested group in a land-cover mix of rubber, oil palm and forest, leading to a decline in their numbers (Luskin et al. 2014). Closer contact between wildlife, livestock and people also increases the danger of zoonoses such as malaria pathogen transmission in the case of wild macaques (Akter et al. 2015; Divis et al. 2015) or rabies from various wildlife species (Wang et al.

2014). Vice-versa, diseases from humans and livestock can spread to wildlife, e.g. infections with *Salmonella* bacteria (Rahman et al. 2005; Singh & Gajadhar 2014).

2.6 Wildlife- friendly farming

Protected areas remain essential refugia for wildlife, especially for forest specialists (Gardner et al. 2009). However, biodiversity conservation also depends increasingly on the extent of conservation in surrounding agricultural landscapes (Daily 2001; Daily et al. 2001; Harvey et al. 2006; Harvey et al. 2008; Baudron & Giller 2014). In areas lacking pristine habitat, other kind of natural vegetation within human dominated landscapes might become a critical shelter for biodiversity (Harvey et al. 2006; Chazdon et al. 2009). Wildlife-friendly farming aims at increasing the heterogeneity in cultivated landscapes through the retention of natural habitat remnants within the plantation or through extensively managed semi-natural landscapes and the reduction in impacts of fertilizers and pesticides (Green et al. 2005; Fischer et al. 2008; Baudron & Giller 2014).While such farming measures can compromise high crop yields in some agricultural systems (Green et al. 2005), it has no noticeable effects in others (Clough et al. 2011). Nevertheless, many wildlife-friendly measures have costs, either direct financial costs or in form of increased labor and require economic incentives for farmers (Häuser et al. 2015).

The land-sparing vs. land-sharing debate is often stuck in an either-or framework which in both extremes is not desirable (Kremen 2015). In the end a synergistic approach is likely to be most successful (Anand et al. 2010; Lentini et al. 2012; Blanco & Waltert 2013; Gilroy et al. 2014; Kremen 2015). As many wildlife species, particularly the larger ones, hardly exceed a certain distance away from larger blocks of natural habitat in their ventures into farmland (Bennett 2003; Maddox et al. 2007; Gubbi 2012; Bernard et al. 2014; Gilroy et al. 2014; Goswami et al. 2014; Yue et al. 2015), wildlife-friendly farming is of high importance in such surrounding areas. Measures that can be implemented in edge habitat zones include reduced input of fertilizer and pesticides and increasing heterogeneity, e.g. in form of reduced weeding and maintenance of understory vegetation or natural vegetation patches. In the case of rubber, a more diversified production, which would also benefit wildlife, can be achieved by intercropping: temporary crop species in the case of young rubber trees such as vegetables or ornamental plants (pers. obs.), permanent plant species in mature plantations with two and more species of e.g., coffee and timber trees, or in

diverse plantations with fruit trees and perennial crops such as ginger (He & Martin 2015). Depending on current socio-economic circumstances, rubber intercropped with tea in Hainan in China seems to be more profitable than only cultivating either crop alone (Guo et al. 2006) and intercropping of rubber with coffee and cocoa in West Africa was found to be more lucrative than rubber monocultures in the first twelve years and not significantly different afterwards (Snoeck et al. 2013). However, crops known to attract wildlife (e.g. maize, rice or banana) should be avoided as mixed crops in areas close to protected land in favor of less palatable species such as tea, chili or ginger to reduce human-wildlife conflicts. Valuable crops such as chili and tea or timber species such as neem or teak can serve as economic incentives for farmers to diversify their rubber system (Somboonsuke et al. 2011). In China, traditionally used medicinal plants that became rare in the wild even have the potential to be integrated into rubber plantations (Häuser et al. 2015). Rubber agroforestry systems ('jungle rubber'), common in Indonesia, support the conservation of many wild plant and animal species (Michon & de Foresta 1995), but are less profitable than intensively managed rubber plantations (Williams et al. 2001; Feintrenie et al. 2010). However, only ceasing of weeding does not seem to have negative impacts on latex yield (Abraham & Joseph 2016) while a more abundant and diverse understory vegetation allows a broader diversity of insects to persist (Meng et al. 2012a; Meng et al. 2012b), which again represent the food basis for mammal species such as bats (Phommexay et al. 2011).

Oil palm seems to be an even poorer biodiversity substitute for natural forests than rubber (Fitzherbert et al. 2008) and potential oil palm agroforestry systems are controversially disputed with strong advocates (Bhagwat & Willis 2008) and strong opponents (Phalan et al. 2009). Mixed tree plantations might be a successful strategy if the palms are not shaded and if the economic output of both crops will be comparable to monocropped systems (Corley & Tinker 2003). Rubber and oil palm mixed systems for example are not compatible as the rubber will cast shade on the oil palms and, hence, reduce yield (Corley & Tinker 2003). As for rubber, an increased heterogeneity of the cropping system is expected to be beneficial for plant and animal biodiversity. The presence of epiphytes and leguminous crops, for example, has positive effects on birds in oil palm plantations (Koh 2008) while keeping or cutting the epiphyte flora did not show any impacts on the crop yield (Prescott et al. 2015). The same applies for the retention of forest patches within oil

palm landscapes, for which no significant negative effects on yield in oil palm were found (Edwards F.A. et al. 2014b).

The Roundtable on Sustainable Palm Oil (RSPO), established by oil palm stakeholders, implemented 39 sustainability criteria aimed to support an environmentally more sustainable production, one being that operations should be managed to ensure that threatened species as well as habitats with a high conservation value are maintained and enhanced (Laurance et al. 2010; Azhar et al. 2015b; www.rspo.org). However, landscape heterogeneity was sidelined in these criteria (Azhar et al. 2015b) and retaining forest remnants might not always automatically lead to increased biodiversity in adjacent plantations as was shown for birds (Edwards et al. 2010). For some mammals on the other side, forest remnants within cultivated land might constitute important "stepping stones" between distant continuous forests as well as shelter habitats, to which they can retreat when migrating or when chased away from plantations. This seems to be mainly the case for widespread species without high conservation concern and overall species richness can be expected to be generally lower in forest patches compared to continuous forest (Bernard et al. 2014).

Ecological requirements of the environment differ between species and conservation management strategies need to adapt to the wildlife assemblages of local conservation concern. So far, species that are frequently recorded in plantations are rarely of high conservation value (Maddox et al. 2007). In the case of bats, a higher number of habitat patches with a broad diversity of plants would likely benefit frugivorous bats through the provisioning of shelter and a more diverse diet (Azhar et al. 2015a) and the maintenance of natural understory vegetation would support a higher abundance of prey insects for insectivorous bats (Fukuda et al. 2009; Shafie et al. 2011). For elephants, a driving factor to access certain areas can be sufficient water availability (Blair & Noor 1981; Sukumar 1989; Chong & Dayang Norwana 2005). By retaining the natural vegetation along streams and around ponds while abstaining from planting highly attractive crops in the vicinity might also reduce conflicts with people.

2.7 Conclusions

Agricultural landscapes in the tropics and subtropics including rubber and oil palm plantations cannot substitute for the high biodiversity found in natural forests (Martin & Sauerborn 2013). Whatever wildlife-friendly measures are implemented, the first purpose of cultivations is still to produce crops for the benefit of a single species, that is, humankind. However, there is potential to facilitate a better co-existence between people and wildlife, from which both sides can benefit. Some species such as wild pigs or leopard cats have already adapted to rubber and oil palm plantations, others such as tapirs are failing and many other species seem to be struggling. While plantations are unable to provide sufficient quality habitat for many forest specialist species (Sodhi & Brook 2006; Wilcove et al. 2013; Yue et al. 2015), it is urgent to support the populations of species with some adaptation potential by implementing wildlife-friendly measures in form of an increased heterogeneity in the agricultural dominated landscape matrices surrounding natural habitats. Some of these measures such as maintaining the epiphyte flora might not result in reduced yield or increased labor, while others such as the retention of forest remnants or natural vegetation patches will need economic incentives for farmers to encourage their implementation.

Although we found more records on mammal species for oil palm plantations compared to rubber plantations in our review, this is likely reflecting the lower number of wildlife assessments available for rubber and overall, rubber plantations seem to have more potential to maintain a higher species richness than oil palm (Fitzherbert et al. 2008). The generally low number of publications assessing mammalian diversity both in rubber and oil palm cultivations shows that there is an urgent need to conduct further studies on species abundance and habitat preferences for wild mammal species under different plantation management regimes.

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Chapter 3

Between conflict and coexistence: Wildlife in rubber-dominated landscapes[2]

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3.1 Abstract

The continuing loss and degradation of their natural habitats forces some wildlife species to increasingly extend their habitats into farmlands, thereby intensifying conflicts with people as resources diminish. Despite massive expansion in rubber (*Hevea brasiliensis*) plantations in recent decades, little is known about the diversity and distribution of wild mammals in rubber-dominated landscapes or the associated human-wildlife conflicts. We assessed the presence and diversity of mammalian wildlife and damage occurrence in such rubber landscapes in southern Thailand, in and around the Tai Rom Yen National Park. We interviewed 180 farmers about wildlife visits to their farms and related damage. We walked 50 transects inside and outside of the natural forest and installed camera-traps at

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the boundary between the plantations and the forest as well as deeper into the forest to assess wildlife presence. A total of 35 mammal species were recorded inside the forest. More than 70% of these were also present at the forest boundary but both species presence and diversity were far lower in the farmland. Elephants (*Elephas maximus*) were responsible for 90% of wildlife damage incidents in rubber and 86% of these cases affected young plants that were not yet tapped. However, while almost half of all the respondents had had elephants visiting their farm, less than half of those reported damage. These results suggest that rubber-dominated landscapes surrounding protected areas have the potential to facilitate coexistence between people and some wildlife species if young plants are better protected and plantation management is made more wildlife-friendly.

Key words: *Human-wildlife conflicts, Asian elephant, Hevea brasiliensis, Thailand*

3.2 Introduction

Conflicts between humans and wildlife are one of the major and persistent challenges to long-term conservation efforts for threatened species (Distefano 2005). In the face of diminishing natural habitats, overlapping resource use by people and wild animals is increasing, thereby creating a continuous source of strife (IUCN World Parks Congress (WPC) 2004). Conflicts are particularly severe at densely populated fringes of protected areas, where chances of contact are high when wildlife enter farmlands in search of forage and water. Common types of conflicts are livestock predation and crop foraging, with some charismatic species being frequently in the focus of complaints (Western & Waithaka 2005; Douglas & Veríssimo 2013). Elephants are one exemplary species which are often mentioned and feared as the damage they cause can be devastating for the affected farmers (Tchamba 1996; Nyhus et al. 2000; Campos-Arceiz et al. 2009). These pachyderms are by far not the only species with a preference for nutritious and easily palatable crops and smaller animals such as rodents might account for considerable damage as well (Lahm 1996; Arlet & Molleman 2010). However, as larger and charismatic species are often protected under national and international laws and extreme damage occurs infrequently but is concentrated in particular localities, farmers feel more helpless and exposed to these animals (Naughton-Treves 1997).

Most food crops planted are savory for both humans and wildlife with some local variations of preferences, also influenced by availability and seasonality. In addition to rice, which seems to be the crop in highest demand by elephants in many places, pachyderms in China (*Elephas maximus*) were reported to prefer wheat if available (Zhang & Wang 2003) while the most targeted crop by their relatives were bananas in Sri Lanka (Campos-Arceiz et al. 2009) and maize in Indonesia (Nyhus et al. 2000). In eastern Kenya, the crops most affected by foraging African elephants (*Loxodonta africana*) were beans and maize (Ngene & Omondi 2009) while in the Democratic Republic of Congo, the most damaged crop was manioc (Inogwabini et al. 2013). Most crops are at risk of being consumed or damaged by elephants but there are nevertheless some crops that show a lower susceptibility to wildlife consumption, for example ginger, tobacco or chili (Danquah et al. 2006; Kioko et al. 2006; Parker & Osborn 2006). The latter is used to produce deterrents in form of projectiles with chili oil extract, smoke or as grease mix on fences (Sitati & Walpole 2006; Le Bel 2015).

While elephants are often blamed for 'attacking' or 'raiding' crops, this is part of their survival strategy in an environment with ever shrinking natural resources (Peterson et al. 2010; Hill 2015). Particularly long-lived species like elephants might continue to use traditional migration routes, even after the surrounding areas have been transformed into farmlands, or are attracted to water resources, both of which can lead to crop damage (Sukumar 1989; Thouless 1994; Sarker & Røskaft 2010). Damage levels, therefore, also depend on the kind and structure of overall cultural landscapes surrounding protected areas (Boafo et al. 2004). While the susceptibility of plantations such as maize, sugarcane or bananas is well documented (e.g., Barnes et al. 1995; Naughton-Treves 1997; Inogwabini et al. 2013), damage to some cash crops such as natural rubber (*Hevea brasiliensis*) might represent collateral damage rather than consumption by elephants (Chen et al. 2013; Chen et al. 2016). However, with increasing demand for latex and the expansion of rubber plantations, the potential for conflicts is likely to increase as well. As of 2013, the global area under rubber cultivation surpassed 10.3 million ha with more than 90% of rubber produced in Asia (FAOSTAT 2016). For farmers in some of these countries such as Thailand and Southwest China rubber has been an important source of income over the last decades and contributes significantly to poverty alleviation (Fox & Castella 2013). The expansion of the area under rubber cultivation has led to the replacement of natural forests and traditional land-use types in these regions (Li & Fox 2012). Based on projected rubber

demand by 2024, a further 4.3 – 8.5 million ha of land will be needed for rubber cultivation (Li & Fox 2012; Warren-Thomas et al. 2015). Severe impacts on biodiversity can thus be expected from the projected expansion in rubber cultivation (He & Martin 2015). Conversely, people cultivating rubber in proximity to natural forests can expect escalation of conflicts with elephants. Over the five years from 2008 – 2012, almost 11000 incidents of damage to rubber plantations by elephants were recorded in 253 settlements in Southwest China (Chen et al. 2016). While there is an insurance scheme in place to compensate affected farmers in this region, it has several shortcomings, one being insufficient funding (Chen et al. 2013). Records of rubber damage by elephants date as far back as 1910-1930 with several thousand pounds of damage reported for Malaysia alone in this period (Hubback 1942) and continuous conflicts in the 1970s, prompting declaration of elephants as a serious pest of rubber (Olivier 1978b). Species other than elephants, which were also reported to damage rubber, include primates such as Thomas' leaf monkeys (*Presbytis thomasi*) or orangutans (*Pongo abelii*) in Sumatra (Marchal & Hill 2009; Campbell-Smith et al. 2012).

Despite the increasing extent of land area covered by rubber, relatively little information is available on conflicts with wildlife in rubber plantations or on the potential level and type of biodiversity of large wild mammals that can be sustained in rubber-dominated landscapes in the long-term without causing substantial damage to rubber trees (Harich & Treydte 2016). Here, we examine the various wildlife species found in rubber-dominated areas and in adjoining natural forests and assess their crop raiding behavior as a basis for developing wildlife-friendly and low risk rubber farming strategies. We selected Thailand for this study because it is the leading rubber producing country with a long history of rubber cultivation (Li & Fox 2012, FAOSTAT 2016).

3.3 Materials and Methods

3.3.1 Study site

The Tai Rom Yen National Park (TRY) is located in the South of Thailand, in the Surat Thani province, between latitudes 8°36' – 8°59' N and longitudes 99°22'-99°37'E. It borders the province of Nakhon Si Thammarat in the East (Fig. 3.1). The Park covers an area of about 400 km^2 at an elevation ranging from around 100 to 1,200 m.a.s.l. TRY was

established in 1991 and includes evergreen forest, partly characterised by limestone formations and associated vegetation types (DNP 2013a; Pfeffer 2013), as well as cultivated landscapes dominated by rubber plantations. The region is a traditional rubber cultivating area (Li & Fox 2012) that lies within the transition zone of the Indo-Burma and Sundaland biodiversity hotspots, home to more than 300 mammal species (Myers et al. 2000). The annual average temperature in Surat Thani is 27.4 °C whereas the average annual precipitation is 1862 mm (Pfeffer 2013).

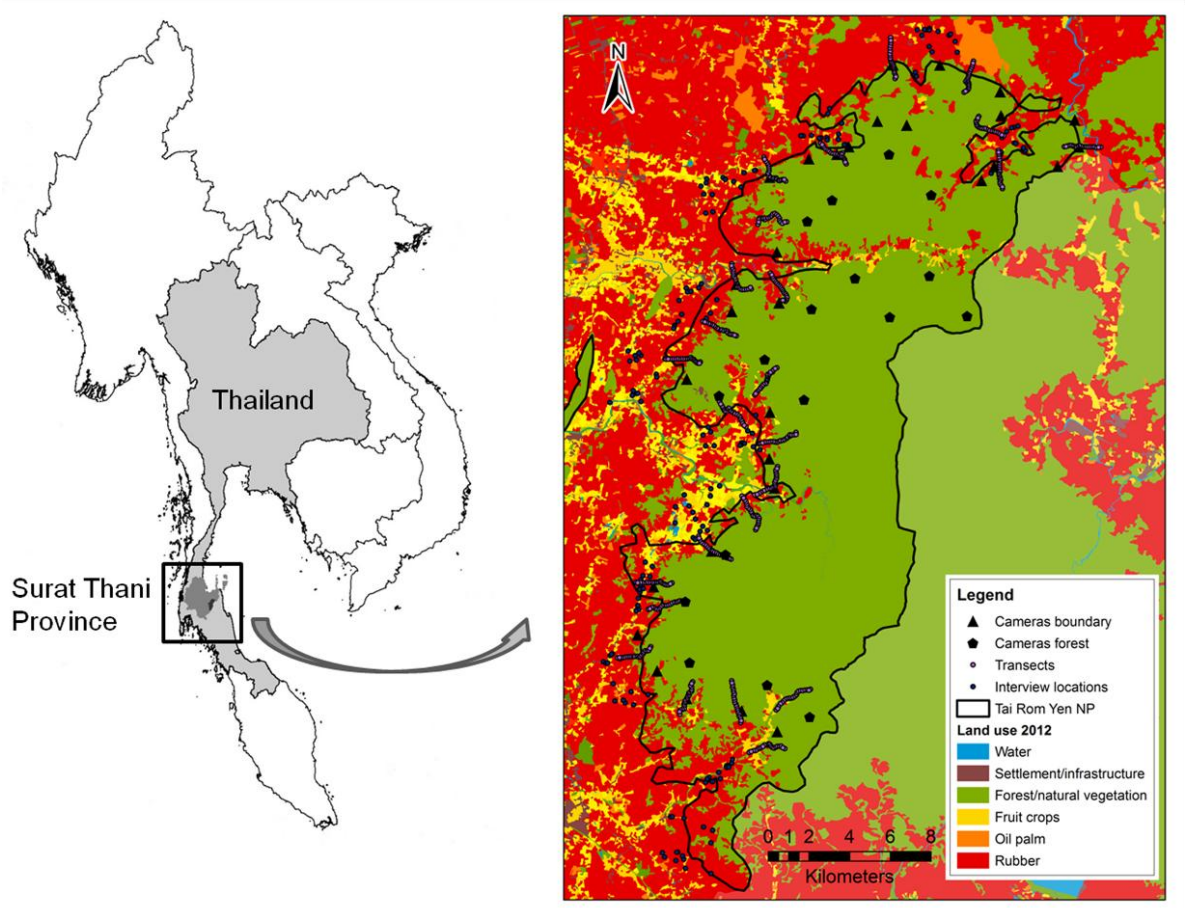

Figure 3.1. The location of the Tai Rom Yen National Park in the Surat Thani Province in Thailand with the associated land-use map provided by the Land Development Department Thailand. The black line shows the boundary of the Tai Rom Yen National Park. Dotted lines are transects, black pentagons are camera locations inside the forest and black triangles are the camera locations at the forest boundary. The forest edge in the map can vary from the actual forest cover due to changes since the map publication in 2012.

3.3.2 Transect surveys

We recorded direct and indirect observations (i.e. dung, footprints, calls and other sounds, feeding signs, scratch marks, burrows, etc.) of wildlife presence on 50, one-km transects in the transition zone of TRY and the surrounding farmland (Fig. 3.1). The 50 transects were arranged into 25 matched pairs, 25 of which were sampled inside the natural forest and the other 25 inside the plantations outside the forest. Each transect started from and was oriented perpendicular to the forest edge as much as practicable. Accordingly, each one-km transect running from the forest edge into the forest was matched with another one-km transect running from the forest edge into the plantation outside the forest. Due to the difficult terrain we could not cut straight transect lines. Instead, we followed wildlife trails in the forest while keeping approximately perpendicular direction to the forest edge (Buckland et al. 2008; Steinmetz et al. 2013). Before the actual data recording, we marked trees every 100 m along the transect line. During the scheduled survey walks we stopped at each of these marks for five minutes to listen to potential wildlife sounds and to record site covariates, i.e. habitat type, habitat structure, elevation, slope, water availability, signs of human presence in the forest and type of crop in the farmland. Each wildlife sign detected on the transect was identified to the species or taxonomic group and the approximate distance of its location to the forest boundary recorded in terms of 100 m intervals. Photos or probes of species that were hard to identify were shown to experts and in case of doubt labeled as unidentified. Each transect was sampled thrice over a period of 16 months spanning the dry and rainy seasons in 2013 and 2014.

3.3.3 Camera trap surveys

Cameras equipped with passive infrared motion detectors and night time infrared illuminators (model Reconyx^{TM} HyperFire^{TM} HC600) were installed in 30 locations along the boundary between the natural forest and rubber plantations over the total duration of 16 months. Criteria for camera locations included accessibility, i.e., spots where chances were high that animals would pass into the farmland, i.e., along wildlife trails or near streams. Where possible, cameras were set right at the forest edge or a maximum distance of 25 m into the forest and were located approximately 1000 – 2000 m apart. A total of 13 cameras were alternately shuffled between the 30 locations. Each location was sampled at least three times with a minimum continuous recoding duration of two weeks, resulting in a

minimum of 60 camera day sampling effort per location. In addition to the cameras set at the forest edge, 21 locations extending up to 3 km inside the natural forest were each sampled once for a minimum recording duration of two weeks. All cameras were mounted on forest trees at an average height of 1.5 to 2 m, depending on the slope and tree, and angled slightly downward to capture the whole range of small to large mammals. Each camera was equipped with a small note explaining the purpose of the survey in Thai language to preempt damage or theft.

3.3.4 Interviews with local farmers

We interviewed 180 farmers around the Tai Rom Yen National Park (TRY) in 2013 about conflicts with wildlife and general wildlife presence on their farms. In the interviews we aimed to determine the extent of conflicts and their impacts on farmers' livelihoods relative to other factors causing crop loss. We further identified the species in the focus of damage complaints and assessed general perceptions towards wildlife conservation and potential benefits from natural resources. Biophysical features such as water availability and individual farm management were included in the evaluation as well. Fifteen respondents were each randomly selected within twelve, 3 km × 3 km blocks around the park boundary. The interviewed farmers inhabited land within a maximum distance of 3 km from the park boundary. The questionnaires were semi-structured with open and closed questions. We provided respondents with pictures from native wildlife species representing major taxonomic groups for identification of those species groups they encountered on their farmland and included alien species photos as controls. If damage had occurred recently then we verified the information given by farmers on site.

3.3.5 Statistical analysis

A generalized linear model assuming a binary error distribution and a logit link function was used to select a subset of nine overall covariates most strongly correlated with the probability of crop damage by elephants from the interview dataset (Table 3.1). These explanatory variables were pre-selected because we considered them as potentially the most likely to influence the probability of elephant damage to crops prior to analysis. With the exception of the two continuous variables 'distance from forest' and 'farm size', all the other variables were categorical. All possible interaction terms between the different

variables plus quadratic terms in the two continuous covariates were also included in the full model. The two continuous variables were internally centered and scaled (standardized) but all parameter estimates and related statistics are reported on the original scale. We selected the covariates most strongly correlated with the probability of elephant damage using the forward selection method, adding covariate effects sequentially. At each step of the selection method covariate effects are chosen and added to the model using the Akaike, corrected Akaike and Schwarz Bayesian information criteria. Selection of effects was made subject to the strong hierarchy (marginality) requirement, meaning that for any interaction term to be included in the model, all main effects that are contained in the interaction term must also be included in the model. Thus; for example, in order for the interaction term region × farm size to enter the model, the main effects region and farm size must also be present in the model. Similarly, neither region nor farm size can leave the model while the interaction term region × farm size is still in the model. We re-ran the model selection process by replacing the categorical covariates, water and crops, with their more detailed derivatives. Model selection was carried out using the SAS GENSELECT procedure (SAS Institute 2016, Version 9.4, SAS/STAT Version 14.1) and SPSS (IBM SPSS Statistics 22).

For analyses of data from the transects, cameras and interviews, we grouped members of the order Artiodactyla and Proboscidea with the overall group of ungulates (see Table B.1 in the appendix for a full species list and classification). Apart from recording the species richness, we further calculated the average presence of the four taxonomic groups of carnivores, rodents, ungulates and primates. Presence or absence was recorded for every 100 m segment along the transects, resulting in 10 binary records (1/0) per transect walk and a total of 30 binary records over the three repeat survey walks per transect. We calculated the average of the 30 records per transect resulting in a value between 0 and 1 and graphically displayed this as the average presence across all the 100 m transect segments for the transect. The average presence for a species in the forest or plantation landscape was then calculated as the average presence across all the 25 transects in each landscape.

Table 3.1. Variables included in the generalized linear model.

Variables	Categories
Region	North Central South
Distance to natural forest (m)	Continuous
Size of farm (ha)	Continuous
Water source	River Stream Pond
Rubber age	Mature Young
Fruit crops	Binary, comprises Durian, Rambutan, Longkong/Langsad, Mangosteen
Diversified farming	Binary (yes / no)
Natural vegetation present	Binary (yes / no)
Prevention measure	Binary, comprises firecracker, guarding of crops, electric fences

3.4 Results

3.4.1 Wildlife presence in the farm-forest transition zone

We recorded a minimum of 35 wild mammal species from 21 families in the farm-forest transition zone through the interviews, transects and cameras. Overall, 35 species were found inside the forest, 25 species at the forest edge and nine species were identified on transects in the farmland, the latter adding up to 20 when including interview data. About 29% of the 35 species are listed as threatened in the IUCN red list (IUCN 2015). During the transect surveys we recorded a total of 25 species while the cameras recorded 26 species in 371 identified pictures taken during a total of 3090 camera trap days and nights.

With 10 species, Carnivora as a taxonomic group displayed the highest species richness, followed by Rodentia (Fig. 3.2). However, due to difficulties in identifying rodent species from the camera trap pictures, signs or actual sightings, this group is likely highly

underrepresented in our recorded species list. Species richness decreased for all taxonomic groups by about 43% on average from the forest to the farmland. This pattern was most pronounced for carnivores for which we recorded a 70% decrease in species richness. At the farm-forest boundary, however, species richness was still relatively high with around 71% of the total number of species recorded. The only exception was shown by primate species, for which we recorded only two species at the forest edge, possibly due to camera positioning focusing on the ground rather than the canopy.

Not only the species richness but also the species presence (expressed as a percent of all transects) was lower in the farmland than in the forest (Fig. 3.3). Average species presence for all groups except humans was 10% lower in the farmland than in the forest. Human presence was highest based on our transect records, even in the forest. In our cameras we recorded 20 occasions where people passed through, and half of these cases could clearly be identified as poaching trips. Excluding humans, rodents had the highest average presence in both the farmland and forest (14%), followed by ungulates (8%) and primates (5%). With only 1%, carnivores had the lowest average presence.

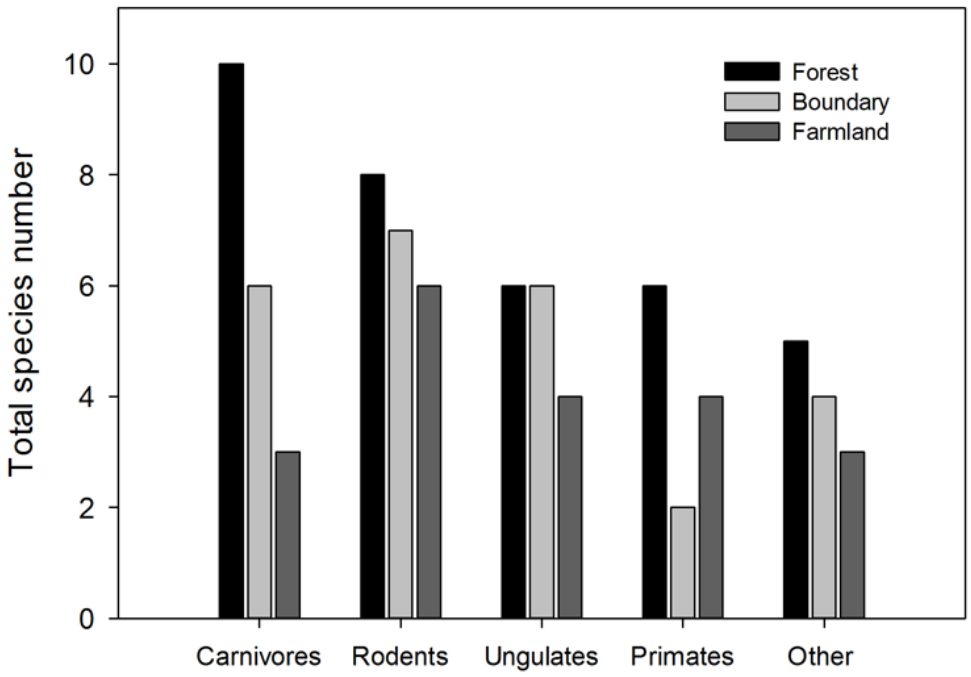

Figure 3.2. Total number of wildlife species belonging to different groups recorded in the natural forest (black bars), at the farm-forest boundary (light gray bars) and in the farmland (dark gray bars) based on data from transects, cameras and interviews combined.

The distribution of the average percentage wildlife presence as a function of distance from the farm-forest boundary varied remarkably among the four species groups but was much higher at almost all distances inside the forest than in the farmland for all the species groups (Fig. 3.4). Only rodents were continuously present up to one km into the farmland (Fig. 3.4), albeit less than 25% as frequent as inside the forest. Ungulate presence decreased strongly after the first 300 m and elephants were the only species, for which signs were found further into the farmland. Also, primate species presence could only be confirmed within 100 m from the forest edge. Carnivore signs were also seldom found to a very low extent in the farmland. Scat, likely from the hog badger (*Arctonyx collaris*), was found up to 800 m in the farmland in an overgrown plantation patch, which had been abandoned for some time.

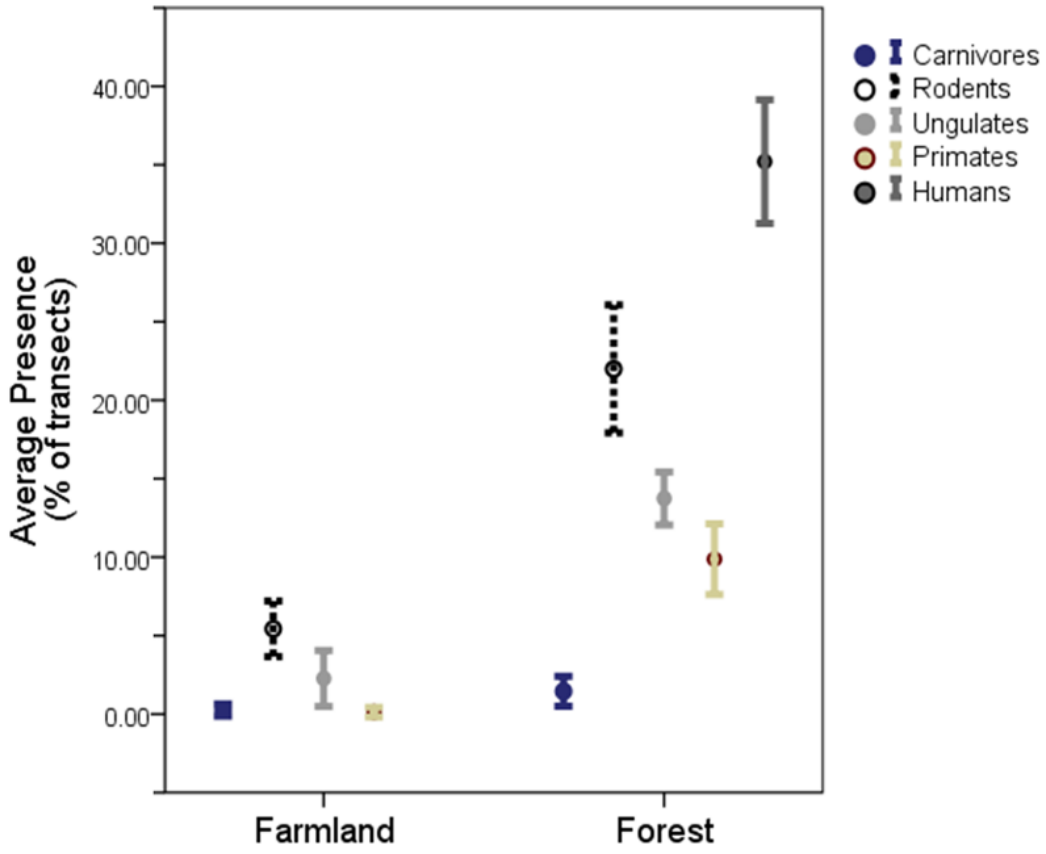

Figure 3.3. Average presence (expressed as percentage of all farm or forest transects) of different wildlife groups in the farmland versus the natural forest (error bars represent 95% confidence intervals). As human presence in the farmland is close to 100%, we displayed human presence for the forest only.

3.4.2 Type and extent of wildlife damage

Farmers identified species from 15 taxonomic groups, which they encountered on their farmlands (Fig. 3.5; Table A1). Crop damage by wildlife affected 40% of farmers. A total of 82 out of 180 interviewed farmers (46%) reported that elephants had been coming to their farms within the six years preceding the interviews. Less than half of those farmers suffered crop damage by elephants (39 interviewed farmers, or 21.7%). Many farmers reported elephant visits, during which the animals were merely eating natural vegetation from the undergrowth or farm fringes and drinking water, if available. Still, elephants were the species most frequently reported to be causing damage (18% of respondents). More than 60% of the farmers who reported species other than elephants as causing most damage had not encountered elephants on their farms. The northern villages around the park were the most susceptible to crop damages by elephants: four to seven times as many farmers experienced crop damage there compared to farmers in the southern and central regions.

The average (± 1SD) size of farms was 4.84 ha (± 3.24) and 97% of the interviewed farmers were cultivating rubber whereas about half of all the respondents were cultivating more than one crop. Crops that were most affected by damage were rubber (67% of all farmers that experienced damage), longkong (*Lansium domesticum*; 41%), durian (*Durio* spp.; 23%) and rambutan (*Nephelium lappaceum*; 15%), partly reflecting the relative abundance of the crops cultivated in the area (Fig. 3.6). Elephants seemed to prefer longkong fruits to other fruits as the ratio of the number of farmers who experienced crop damage to the total number of farmers cultivating the various crops was 1.8 times higher for longkong than for rubber. Elephants were nevertheless responsible for 90% of damage incidents in rubber plantations. Damage to rubber affected young trees (below approximately seven years of age) in 86% of these cases.

The best logistic regression model relating the probability of elephant damage to crops selected by the forward selection method included region, distance to the forest edge, presence of natural vegetation, young rubber, presence of a water pond, farm size and the interaction of farm size with natural vegetation as the best supported and significant explanatory variables (Table 3.2).

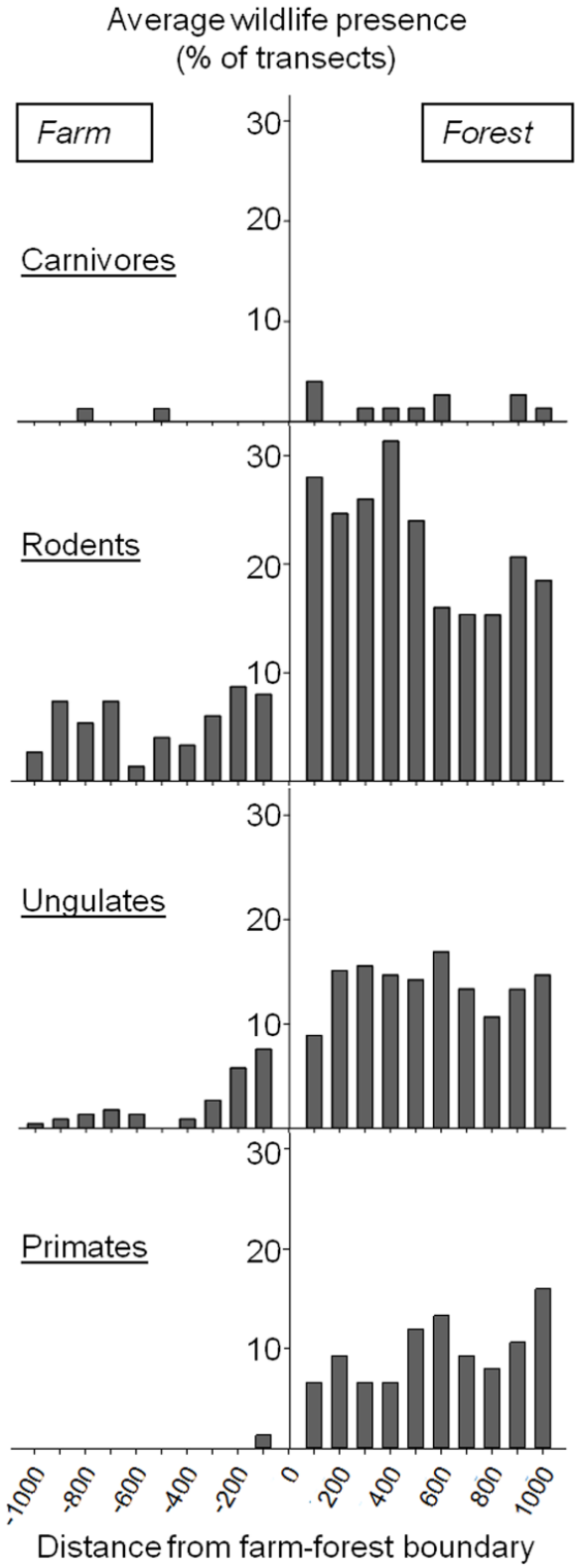

Figure 3.4. Average presence (in % of transects) of different wildlife species or groups recorded as a function of distance from the farm-forest boundary (0 represents the forest edge).

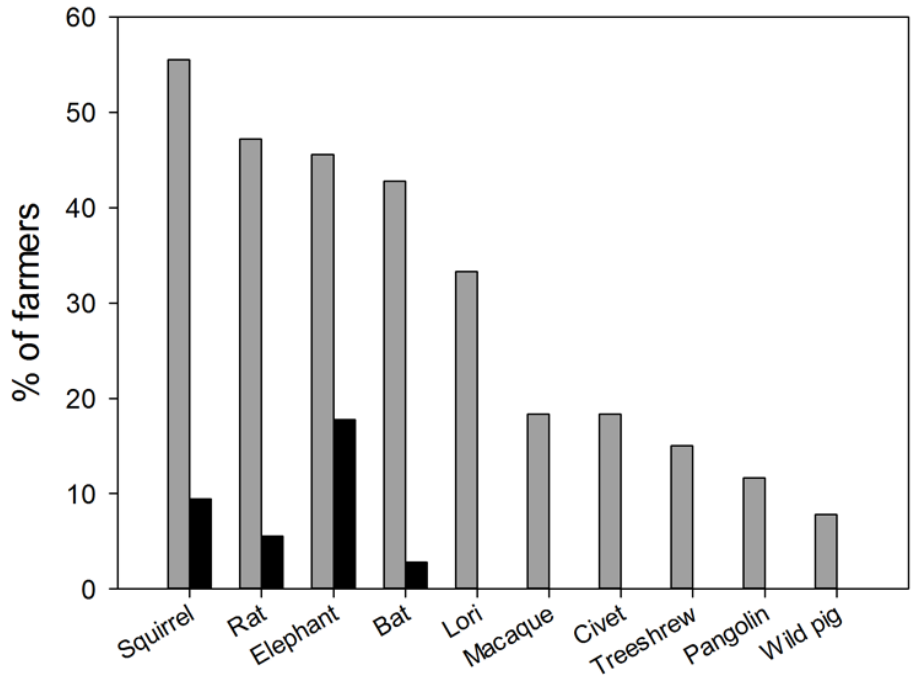

Figure 3.5. The top ten wildlife species or groups encountered by the respondents on their farms (gray bars; % of interviewed farmers) and the species reported to be causing most damage (black bars).

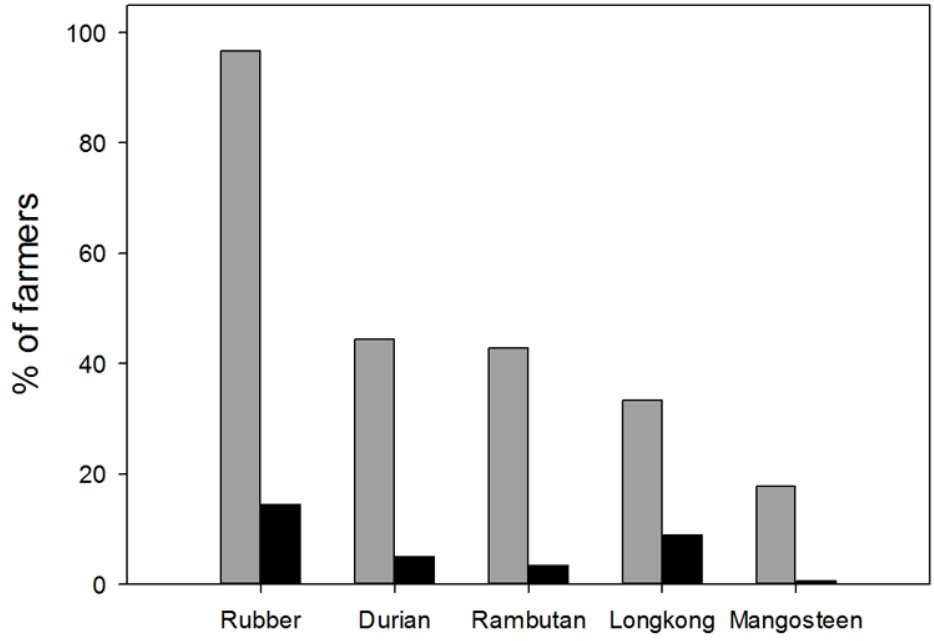

Figure 3.6. Crops cultivated in the study region (gray bars; % of respondents) and the proportion of farmers experiencing damage by elephants for each crops types (black bars).

Table 3.2. Logistic regression model relating the probability of elephant damage to crops with predictor variables.

Variable	Estimate	S.E.	*P*	Exp(Estimate)	95 % CI for Exp(Estimate)	
					Lower	Upper
Region South	0.000	0.000				
Region North	1.736	0.556	0.002	5.677	1.910	16.876
Region Central	-0.998	0.813	0.220	0.368	0.075	1.814
Distance to forest	-0.001	0.001	0.019	0.999	0.998	1.000
Natural vegetation	0.242	1.224	0.843	1.274	0.116	14.043
Young rubber	1.140	0.500	0.022	3.128	1.175	8.329
Pond	2.148	0.950	0.024	8.567	1.332	55.111
Size of farm	-0.120	0.200	0.547	0.887	0.600	1.311
Size of farm * natural vegetation	0.416	0.227	0.067	1.515	0.971	2.364
Constant	-3.210	1.226	0.009	0.040		

The model explained 54% (Nagelkerke r^2) of the variation in the probability of crop damage incidences and correctly classified 85.6% of the observed cases. Damage by elephants was 5.7 times more likely to occur in the North of the National Park than in the South after controlling for variation in the other explanatory variables. The presence of a young rubber plantation increased the likelihood of damage threefold when all the other explanatory variables were kept constant. Farmers with a water pond on or close to their farmland were 8.6 times more likely to experience crop damage assuming constancy in the other variables. While the main factors 'natural vegetation' and 'size of farm' were not significant predictors of the damage probability, a unit increase in farm size was associated with a 52% increase in the likelihood of crop damage where natural vegetation was present than where it was absent. Further, distance to the natural forest was a significant predictor of the probability of crop damage (P = 0.019) but within the range of the three km distance we considered, a unit increase in distance from the forest edge was only accompanied with a slight reduction in the likelihood of crop damage.

3.4.3 Farmers' perspectives on crop damage

With only about one fifth of farmers experiencing crop losses by elephants around the National Park, wildlife damage to crops was not among the major concerns of the farmers in the region. Most farmers mentioned crop losses due to diseases (e.g. fungi) as the major reason for crop damage (80.6% of farmers), followed by insect damage (13.3%) and only 7.8% considered wildlife, and particularly elephants, as the major threat to their crops. Nevertheless, elephants sometimes caused substantial damage to crops in individual farms. Notably, elephants hardly caused damage in mature rubber plantations (only 15% of all damage events for rubber) and if damage occurred, the trees were usually able to recover (e.g. broken branches). This is significant as 96.7% of the interviewed farmers were cultivating rubber. Consumption of rubber bark by elephants as observed in other areas of Thailand (Dr. Mattana Srikrachang, personal communication) or in Malaysia (Blair 1981) was not observed or reported from our study site.

Attitude towards elephants was generally positive, with 68% of farmers stating they liked the pachyderms or at least liked them if they were not causing damage. Only 20% of the farmers stated a clear dislike for the animals, a feeling significantly related to the perceived danger posed by elephants (χ^2 = 7.735, df = 1, P = 0.005) but not to any actual crop damage experience by individual farmers (χ^2 = 0.194, df = 1, P = 0.660).

3.5 Discussion

3.5.1 Presence and detection of wildlife

Decreases in biodiversity are usually observed in farmlands compared to natural forests as was the case for our study area (Koh 2008; Gilroy et al. 2014; Yue et al. 2015; Crisol et al. 2016). Within a 2 km transition zone from natural forest to farmland, we observed a decline in mammalian species diversity of almost 75%. Including data from interviews within 3 km outside of the forest, the decline was still more than 40%. Ground-truthing of information obtained through interviews has been recommended in ecological surveys and the combination of transect, camera and interview surveys complemented our overall data collection well (White et al. 2005). In the case of carnivores, cameras rather than transects might be more effective tools for detection due to their relative rarity, nocturnal and often secretive and elusive behavior (Karanth et al. 2004; Chutipong et al. 2014). This might

partly explain the relatively low presence of carnivores in the farmland where we did not install cameras due to frequent human activity and the risk of theft. In contrast, transect surveys proved to be more efficient in detecting primate species as we only recorded two out of four species with the cameras. Arboreal species like gibbons (*Hylobates lar*) and loris (*Nycticebus coucang*) were recorded during transect surveys but were not detected by the cameras likely because we mounted cameras at a slight angle to the ground. The rodent species richness we recorded were comparatively few, reflecting the difficulty of identifying animals down to species level from camera trap pictures or from direct sightings on transect walks.

3.5.2 Factors influencing crop damage

Overall conflict levels were comparatively low, also for elephants, as less than a fifth of farmers experienced elephant crop-foraging within a 3 km radius around the natural forest. Conflicts also decrease with increasing distance from the natural elephant habitat, partly explaining why the aggregate crop loss through elephants is usually negligible at the national scale (Barnes et al. 1995; Naughton-Treves 1997; Naughton-Treves & Treves 2005; Sarker & Røskaft 2010; Wilson et al. 2015; Chen et al. 2016). However, in close proximity to the forest boundary the individual damage experience can be very high, consistent with the typically localized but severe crop loss through elephants (Naughton-Treves 1997; Naughton-Treves & Treves 2005; Nath et al. 2015). The northern district around TRY was particularly affected by conflicts and hence the region was unsurprisingly a significant predictor of the probability of damage, with damage levels higher for the northern than the other parts of the park. One village in the North, surrounded on three sides by forest cover, was particularly affected by elephant foraging. Still, only one third of farmers in that village ranked wildlife as the most important cause of damage to their crops. Across the entire surveyed area even fewer respondents (8%) ranked wildlife as the topmost cause of crop damage and the vast majority of the respondents worried more about plant diseases as the leading cause of crop damage (Thirgood et al. 2005). Two reasons might account for the relatively low level of overall crop damage by wildlife. First, no exact records of elephant numbers are available for TRY but the population is likely rather small. The elephant population size for the overall forest complex to which TRY belongs was estimated to be less than 100 animals (Srikrachang 2013), and we assume there might be less than 50 elephants in TRY itself based on our field observations and records.

Second, the land-cover was dominated by rubber trees, which were hardly susceptible to damage by elephants once they reached maturity and had strong enough stems at around five to eight years of age. A similar observation was also made for rubber in China where less than 1% of damaged rubber trees were older than 8 years (Chen et al. 2013). As more than 45% of farmers had elephants visiting their plantations and elephant damage occurred in less than half of these cases, the low damage levels cannot be attributed entirely to small elephant numbers but are more likely attributed to the fact that damage was mainly restricted to young rubber plants. Elephants were reported to feed on rubber seedlings but a high proportion of damage to seedlings and saplings might occur through trampling or other destructive behavior when elephants pass through a rubber plantation (Olivier 1978b; Blair & Noor 1981; Zhang 2011; Chen et al. 2013). The damage in young rubber plantation can lead to economic losses due to delayed latex harvest and replanting costs, which averaged about 126 US$/ha and represented a mean loss in rubber of 2.6% per affected village in Southwest China (Chen et al. 2013). While the monetary loss in the case of rubber as a cash crop is relatively high, the proportional loss of some crops in food crop plantations can be even much higher. For example, the potential yield lost because of elephants was 20% for maize and 25% for sorghum in some villages in India in 1981-82 (Sukumar et al. 1998). On the farm level, 16 to 31% of rice, groundnut and millet fields were damaged by elephants in Ghana during 2000-2002 while 28 to 38% of banana, maize, and bean fields were damaged in Uganda in 1996-97 (Adjewodah et al. 2005; Chiyo et al. 2005). In our study site in southern Thailand, Longkong fruits seemed to be more preferred by elephants as the level of damage they suffered was high relative to the number of farmers cultivating this crop. The spiky and strong smelling Durian fruits were favored as well, an observation that was also made more than 70 years ago in Malaysia (Hubback 1942). Fruits are generally a welcome addition to the diet of Asian elephants and in the case of African forest elephants (*Loxodonta cyclotis*) even a strong spatial association between trails and fruit trees was found in the equatorial forests of the African continent (Blake & Inkamba-Nkulu 2004; Jothish 2013; Sekar & Sukumar 2015a; Sridhara et al. 2016). While cultivating different crops on a farm increased the risk of elephant damage in Ghana (Sam et al. 2005), a greater variety of crops within one farm had no significant influence on damage probability in our logistic regression model, possibly because the major crop in the area was rubber. Despite the high proportional damage to Longkong trees, they did not significantly explain the variation in damage probability in our model. Presence of ponds, in contrast, tremendously increased the risk of damage to crops and

other assets in our study area, with one farmer losing almost his entire fish stock when elephants took a bath in his fish pond in 2014. Water sources are well known to be highly attractive to elephants, particularly during dry seasons, and can therefore be expected to elevate the risk of elephant damage to nearby plantations (Thouless 1994; Naughton et al. 1999; Bal et al. 2011).

3.5.3 Farmers' attitude towards elephants

Wildlife species other than elephants were hardly considered by farmers as causing substantial damage, particularly in the case of rubber plants. As the vast majority of farmers cultivated rubber as a major crop, damage to this cash crop might be perceived more critically than the continuous but low-level consumption of other crop types by small mammals (Decker & Brown 1982; Daley et al. 2004; Arlet & Molleman 2010). Nevertheless, the majority of farmers had a positive or conditionally positive attitude towards elephants, which we expected to be associated with the overall low level of damage. However, attitudes seemed to be unrelated to the actual damage experience as only a fourth of farmers stating a dislike of elephant suffered crop losses. In contrast, dislike of elephants was associated with fear, which might be a result of the prevalent latex harvest procedure. Rubber trees are tapped in the night or early morning hours when it is dark and accidental encounters with elephants pose great risks to farmers. Sometimes, elephants stayed on or visited some plantations for several days, forcing farmers to refrain from tapping. The relatively high tolerance for economic outfalls due to elephants in Thailand might also reflect a cultural heritage. Elephants are highly revered in Thailand and are believed to bring good luck (Ringis 1996; McNeely 2000). The killing of elephants is therefore a cultural taboo in Thailand whereas it is seen as a possible management option by many people living with elephants in Africa (Taylor 1993; Hoare 2001; Gadd 2005; Harich et al. 2013). The relatively high acceptance of elephants by the Thai population around TRY did not in itself reflect support for the National Park. Many farmers had been in the area when TRY was established in 1991 and have felt burdened by the restrictions imposed on them following the establishment of the park, particularly as TRY was a stronghold of communist insurgents in the 1980s (DNP 2013a). The animosity towards these restrictions has not completely ceased and the currently high poaching levels can partly be regarded as continuing acts of defiance by some aggrieved groups in the local population.

3.5.4 Management strategies for human-wildlife coexistence in rubber-dominated areas

The underlying problem of conflicts between people and wildlife is the continuous loss or degradation of wildlife habitat and, in the case of TRY, the only remaining natural habitat left is in a mountainous area too unattractive to be converted to farmland. To enrich this restricted habitat and make it more attractive for wildlife to stay within the forest, more artificial water sources and salt licks could be established; a recent government project in TRY is addressing these and related issues focusing on habitat improvement (Weir 1972; Zhang & Wang 2003; Chandrajith et al. 2009; DNP 2013b). The project also involved planting preferred elephant tree species in the forest as decline in such food resources might lead to dispersal of elephants to areas outside of the protected area, thereby escalating conflicts (Osborn 2002). To stop elephants from entering their farms, several farmers have erected electric fences around their farms but have not installed any fence controllers, thereby creating electric currents lethal to elephants. Electrocution is thus a serious threat to elephants, and has caused the death of 118 elephants in India over the 12 years spanning 2001 to 2012 (Palei et al. 2014). Around TRY, electric fences were responsible for the deaths of two elephants in two separate incidents in 2013. These deaths attracted serious penalties for the concerned farmers. Installing simple and cheap fence controllers and electric fences would generally provide effective protection to crops from elephants but their proper design and maintenance are challenging and thus often undermine their long-term success (Sukumar 1989; Hoare 2012). Young rubber plantations could also be fenced off until the risk of damage has substantially decreased but this should be accompanied with awareness of how to set up and maintain non-lethal fences correctly. Another option could be to set up chili grease fences, which have been successfully applied in parts of Africa (Sitati & Walpole 2006). The effectiveness of such fences might however be limited in humid environments where chili-grease would need to be applied more frequently (Chelliah et al. 2010). Because elephants do not specifically target young rubber plantations, a small nuisance might theoretically be sufficient to prompt elephants to initiate alternative routes in the farmland. While chili-grease fences in combination with community-based guarding were successful in keeping elephants from fields in Indonesia, the effects were not more successful than traditional methods like noise in combination with guarding (Hedges & Gunaryadi 2010). In the case of fruit crops or cereals, for which harvests are more seasonal than for rubber, farmer collaborations for guarding crops were

also found to significantly reduce damage in other areas of Asia (Nath et al. 2015). This could also be an option for those farmers around TRY cultivating fruits, but would require good collaborative engagement at the community level. As fruit crops are more susceptible to elephant induced damage, rubber could be planted as a buffer crop within the first one km from the forest, given young plantations can be protected sufficiently.

Whatever measures are taken to reduce loss in crops, some damage will inevitably occur wherever humans and elephants live in close proximity. Insurance schemes can help to make these losses tolerable for affected farmers if the reimbursement is fair and timely and does not prompt farmers to reduce their individual risk-averting management (Nyhus et al. 2005). In China, where an insurance scheme reimburses the loss of rubber trees, the payout system was not adequate and did not account for temporal nor spatial differences in risks (Chen et al. 2013). Even though farmers were not satisfied with the performance of the insurance scheme, the majority nevertheless opted for improved compensation as a measure to mitigate conflicts (Zhang 2011; Chen et al. 2013). Chinese authorities even started to establish crop fields whose only purpose is to distract elephants from the fields of local residents (Luo 2007; Lin et al. 2011). To increase the resources available to elephants, corridors connecting fragmented natural areas should be established (Lin et al. 2008; Areendran et al. 2011). In the Northeast of TRY, for example, a corridor could be established to link TRY to the Sikiet Waterfall National Park, as the two protected areas are located less than 500 m apart from each other and the scarcely populated area separating them is not too steep (Mañas 2015).

3.5 Conclusions

The acceptance of wildlife in close proximity to human activities ultimately depends on how much the life and livelihoods of the people are affected. In the case of the rubber-dominated landscape around TRY, damage to rubber was restricted almost exclusively to young plants. This provides the opportunity to develop protection measures to reduce these collateral losses. The consumption of rubber tree bark by elephants to meet potential nutritional deficiencies might occur occasionally but is unlikely to be a major problem in mature plantations as this type of damage seems mainly confined to young plants (Olivier 1978b; Blair 1981). One key strategy for ensuring peaceful coexistence between people and elephants in rubber-dominated areas, therefore, lies in the protection of young trees.

Another challenge for the long-term conservation of wild mammals in areas with limited natural habitats such as TRY is how to enable wildlife to extend their resource utilization into cultivated lands without causing damage to crops, e.g., by leaving some natural vegetation for wildlife to feed on and for cover on the farms and hence soften the forest-farmland edge. More than 70% of forest wildlife species were found close to the forest-farmland boundary in our study area. This gives some hope that, given a more wildlife-friendly and low risk plantation management strategy, rubber farmlands outside protected areas may serve as extended habitats for some wild mammals. However, more research is needed to guide the development of a set of wildlife-friendly measures that would neither substantially lower the economic output of plantations nor significantly increase the work load of farmers.

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Chapter 4

Seed dispersal potential of Asian Elephants[3]

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4.1 Abstract

Elephants, the largest terrestrial mega-herbivores, play an important ecological role in maintaining forest ecosystem diversity. While several plant species strongly rely on African elephants (*Loxodonta africana; L. cyclotis*) as seed dispersers, little is known about the dispersal potential of Asian elephants (*Elephas maximus*). We examined the effects of elephant fruit consumption on potential seed dispersal using the example of a tree species with mega-faunal characteristics, *Dillenia indica* L., in Thailand. We conducted feeding trials with Asian elephants to quantify seed survival and gut passage times (GPT). In total, 1200 ingested and non-ingested control seeds were planted in soil and in elephant dung to quantify differences in germination rates in terms of GPT and dung

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Harich, F.K., Treydte, A.C., Ogutu, J.O., Roberts, J.E., Savini, C., Bauer, J.M. & Savini, T. (2016). Seed dispersal potential of Asian Elephants. *Acta Oecologica, 77*, 144-151. doi:10.1016/j.actao.2016.10.005
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treatment. We used survival analysis as a novel approach to account for the right-censored nature of the data obtained from germination experiments. The average seed survival rate was 79% and the mean GPT was 35 h. The minimum and maximum GPT were 20 h and 72 h, respectively. Ingested seeds were significantly more likely to germinate and to do so earlier than non-ingested control seeds (P = 0.0002). Seeds with the longest GPT displayed the highest germination success over time. Unexpectedly, seeds planted with dung had longer germination times than those planted without. We conclude that *D. indica* does not solely depend on but benefits from dispersal by elephants. The declining numbers of these mega-faunal seed dispersers might, therefore, have long-term negative consequences for the recruitment and dispersal dynamics of populations of certain tree species.

Key words: *Dillenia indica, Elephas maximus, seed germination, survival analysis, Thailand*

4.2 Introduction

With ongoing forest fragmentation and losses, the seed dispersal of some tropical plants is becoming increasingly hampered as populations of large seed dispersal agents are declining and their movements are being restricted (Corlett 2002). This is of concern for overall forest diversity as the dispersal of seeds away from the parent organism is an essential strategy used by plants to find suitable establishment sites of reduced competition, herbivore or pathogen attacks (Howe & Smallwood 1982; Harms et al. 2000; Willson & Traveset 2000; Corlett 2014). Dispersal mechanisms include abiotic drivers such as wind or water and biotic dispersal modes such as endo- or epizoochory, with vertebrates as dispersal agents (van der Pijl 1972; Burrows 1986; Murray 1986; Fleming & Kress 2011). A broad range of different animal species can serve as seed dispersers, including birds, bats, rodents, carnivores, primates and terrestrial herbivores (Howe 1986; Stiles 2000; Corlett 2014). Provided the seeds can survive the consumption process, frugivorous animals, particularly the large-sized animals, can disperse seeds over wide distances (Seidler & Plotkin 2006). Among large herbivores, elephants are noteworthy in playing a prominent role in maintaining tree diversity in forest ecosystems. With a diet comprising more than 350 different plant species, African forest elephants (*Loxodonta cyclotis*) consume the broadest spectrum of fruits of all extant elephant species (Blake 2002) while Asian elephants (*Elephas maximus*) reportedly forage on around 100 different

plant species (Sukumar 1989; Chen et al. 2006; Campos-Arceiz et al. 2008a; Baskaran et al. 2010; Campos-Arceiz & Blake 2011).

Hence, the range of plant species consumed by elephants varies greatly across geographic regions as do their daily travel and, therefore, potential seed dispersal distances (Sukumar 1989). Forest elephants in Ivory Coast have been reported to cover 1-15 km / day, for an average of about 6 km / day (Theuerkauf & Ellenberg 2000) whilst in northern Congo their travel distance varied between 2 and 22 km / day (Blake 2002). However, the actual distances over which elephants can disperse seeds can be much larger, especially for large seeds, which can take several days to pass through the digestive tract (Powell 1997). Notably, travel and dispersal distances of up to 57 km over a period of three days have been recorded for elephants in the Congo (Blake et al. 2009). The maximum dispersal distance for Asian elephants varies with geographical conditions and can range from an estimated 4 - 6 km in Myanmar and 46 - 54 km in India, with 50% and >80% of seeds being dispersed over 1 km distances from their origins, respectively (Campos-Arceiz et al. 2008b). This implies that both African and Asian elephants could potentially disperse seeds over distances as large as 54 - 57 km. In tropical forests such distances are much larger than the maximum dispersal distances of other seed dispersers. Distances can be more than seven times longer than the maximum dispersal distance for black-casqued hornbills (*Ceratogymna atrata*) in West Africa and about 43 times longer than the maximum recorded dispersal distance for gibbons (*Hylobates mulleri x agilis*) in Borneo (Holbrook & Smith 2000; McConkey 2000; Blake et al. 2009). Asian elephants might, therefore, rank among the most important long-distance seed dispersal agents in Asia (Campos-Arceiz et al. 2008b).

Some trees have even adapted to this mode of dispersal, the so-called "megafaunal-syndrome" (Janzen & Martin 1982; Guimarães Jr. et al. 2008; Blake et al. 2009; Campos-Arceiz & Blake 2011). Dispersal syndrome refers to a general set of characteristics of fruits and seed traits which are associated with a particular mode of dispersal, e.g. the evolvement of large fruits and seeds that attract megafauna as consumers and dispersers (van der Pijl 1972; Howe 1985). Several plants such as *Balanites wilsoniana*, *Sacoglottis gabonensis*, *Irvingia gabonensis* and *Panda oleosa* likely rely exclusively on African forest elephants as seed dispersal agents for spatial distribution, increased germination success and reduced germination time with associated reduced exposure to seed predators (White

1994; Cochrane 2003; Babweteera et al. 2007; Blake et al. 2009;Campos-Arceiz & Blake 2011). In contrast, no such obligate seed dispersal mutualism has been recorded for Asian elephants thus far and they seem to disperse fewer seeds from fewer tree species than their African forest elephant counterparts. This view might however be biased due to the overall poorer knowledge of Asian elephant nutritional ecology (Corlett 1998; Kitamura et al. 2007; Campos-Arceiz et al. 2008b; Corlett 2014). While the passage of seeds through the gut of an African elephant generally enhances germination probability, there is little comparable data for the Asian elephant. One experimental study that explored the influence of gut passage on seed germination in the Asian elephant was disturbed too early to draw firm conclusions (Kitamura et al. 2007) whilst a second study found negative effects for tamarind (*Tamarindus indica)* seeds after ingestion (Campos-Arceiz et al. 2008b). In the face of declining numbers of large mammals in Southeast Asia (Ripple et al. 2015), more insights into their importance for the dispersal of seeds of different tree species are necessary to assess threats to forest ecosystems. Results from Africa showed that the loss of elephants (and other large frugivores) negatively affects the recruitment of animal-dispersed tree species, thereby fostering the development of species-poor tree communities with abiotic dispersal modes (Blake et al. 2009). Animal-dispersed tree populations in contrast will likely face increased clustering, contraction of their geographic ranges and reduction in genetic variation if the numbers of their dispersal agents decline or vanish altogether (Cramer et al. 2007; Terborgh et al. 2008; Markl et al. 2012).

Also in Southeast Asia, defaunated forests are very likely to face declines in tree diversity over time (Pérez-Méndez et al. 2015). Large frugivores like tapirs (*Tapirus indicus*) can be effective dispersers for small-seeded plants but seem to be only limited substitutes for megafaunal seed dispersers (Brodie et al. 2009; Harrison et al. 2013; Caughlin et al. 2014). Even so, few detailed studies have experimentally tested the impacts of Asian elephant fruit consumption on seed dispersal efficiency and studies of their frugivory and seed dispersal potential are still rare (Campos-Arceiz et al. 2012; Sekar et al. 2015b). However, Sekar et al. (2015b) recently assessed the potential of domestic bovids as replacements for elephant seed dispersal in India and Sekar et al. (2013) investigated the ecology of *Dillenia indica*, which is known to be eaten by elephants.

We expand upon the studies of Sekar and Sukumar (2013) and Sekar et al. (2015b) by using *Dillenia indica* as an exemplary megafaunal syndrome species to empirically (*i*)

establish whether and to what extent the seeds survive gut passage, (*ii*) assess if the seeds that have passed through the elephant gut have a higher average germination rate than control seeds that have not, (*iii*) assess the effects of planting ingested and control seeds with or without elephant dung, and (*iv*) quantify the degree to which the gut passage time (GPT) affects the viability of seeds. With this study we also aim to highlight the importance of seed dispersal for overall forest diversity and general biodiversity conservation in the context of land-use changes.

4.3. Materials and Methods

4.3.1 Study site

The feeding and germination experiment was conducted in northern Thailand, in cooperation with the Golden Triangle Asian Elephant Foundation (GTAEF), located in the border area between Thailand, Myanmar and Laos (Corlett 1998; Kitamura et al. 2007; Campos-Arceiz et al. 2008b; Corlett 2014). The annual precipitation is about 1550 - 1650 mm with a peak from June to September and a dry season from December to March. The average daily temperature ranges from 25.8 °C to 27.7 °C (unpublished GTAEF records). The natural vegetation of Northern Thailand is characterized by a mosaic of evergreen and deciduous forest patches (UNODC 2006; Chin 2009). Elephants of the foundation are ex-street begging elephants rescued to a forest environment in Northern Thailand. They are partly kept in disturbed natural forest remnants, partly on grasslands in the floodplains of the Ruak river, a tributary to the Mekong river, and partly in open barns. The animals are sometimes used for touristic activities like riding and bathing, for an approximate average of 3.5 hours and a maximum of 5 hours per day. For most of the remaining time, elephants are allowed to roam in the forest or grassland, but are restricted by up to 30 m long chains in the night.

4.3.2 Study species

Dillenia indica L. is an evergreen tree species of the family Dilleniaceae found throughout South and Southeast Asia, including the natural habitats of the Asian elephants (Gardner et al. 2000). The tree can grow up to 30 m in height (Van Steenis 1948; Abdille et al. 2005; Sekar & Sukumar 2013). Its fruits are large, around 10 cm in diameter, with many small

seeds of about 6 mm in length that are protected by a hard mesocarp (Van Steenis 1948; Gardner et al. 2000). No significant arboreal frugivores were observed for *D. indica;* rodents as well as rhesus macaques (*Macaca mulatta*) are generally unable to access the seeds but some bovids (e.g. gaur *Bos gaurus*) can consume the fruits and seeds (Sekar and Sukumar 2013; Sekar et al. 2015b). However, some individuals have difficulties dealing with the hardness of the mesocarp and elephants were found to eat more than twice as many fruits as the wild and domestic bovids combined (Sekar and Sukumar 2013). As the species is often found at watersides, it is not clear how much it relies on elephants relative to water for its seed dispersal (Van Steenis 1948; Abdille et al. 2005; Sekar & Sukumar 2013).

4.3.3 Feeding trials

We selected six female elephants for our feeding trials, ranging in age from 6 to 35 years and in body weight from 2.9 to 3.5 tons. All elephants were born in captivity with the exception of the oldest one, for which no data were available. Elephants were seasonally allowed to range in a nearby forest with some restrictions, but not at the time of our experiments. Their normal diet of mainly grasses and various other feeds (e.g. bamboo, sugarcane, bananas) was maintained during the feeding trials. The animals were regularly checked by the foundation's veterinarian and were in good health. We offered the animals ripe *D. indica* fruits ad libitum. The elephants were fed one at a time to facilitate a detailed monitoring of their defecation time and to ensure enough manpower was available to retrieve all the dung and seeds. Before being fed to the elephants, the fruits were weighed and the number of seeds they contained estimated based on a regression model of seed number vs weight of control fruits (Van Steenis 1948; Datta & Rawat 2008; Sekar & Sukumar 2013). We monitored the elephants throughout the day and sieved the collected dung through a 2-mm wire mesh with water hoses. In the evenings, elephants were brought to a barn or to resting grounds. We collected the dung defecated in the nighttime in the early morning and assigned all seeds retrieved the mean time between when we stopped and resumed monitoring. We then dried the collected seeds and stored them in labeled paper envelopes for planting within one week of their collection date. We stopped dung collection when no further seeds were found in the dung over the course of at least 12 consecutive hours.

4.3.4 Germination trials

We counted seeds extracted from elephant dung and planted them in 2l pots with commercial potting soil at a nursery shaded with shadow nets. Five seeds were planted per pot and pots were regularly watered. We sequentially planted the seeds retrieved from the different study animals to minimize any potential negative effects of prolonged seed storage time on their germination ability. As the gut passage time (GPT) as well as the deposition of seeds in dung can impact seed survival and seedling growth (Campos-Arceiz et al. 2012) we included the two treatments 'GPT' and 'dung' in our germination experiments. For the GPT treatment, we assigned the seeds to different GPT categories to assess the effect of GPT on the germination rate or time to germination. We selected four categories: one for control and three GPT categories, according to the time of peak seed retrieval and whether the levels of seed loads in the dung piles were increasing or decreasing. The four categories were delineated as follows (*i*) control: fresh and non-ingested control seeds, (*ii*) short: all seeds retrieved within 30 h of GPT (GPT $\leq$ 30 h; n = 1878), (*iii*) medium: all seeds retrieved after 30 h but within 48 h (30h < GPT $\leq$ 48 h; n = 3797), and (*iv*) long: all seeds retrieved after 48 h (GPT > 48 h; n = 581). For each of the six elephants, we planted a total of 200 seeds, 150 divided into the three GPT categories plus 50 fresh and non-ingested control seeds. In aggregate, 1200 seeds were planted. A total of 300 seeds were planted for the first GPT category plus another 300 seeds for the control treatment. We planted 410 seeds for the second GPT period while for the last GPT category, only 190 seeds were available due to the fast digestion of some elephants. Half of all the seeds planted in each of the three GPT categories were planted in pot soil only and the other half in combination with elephant dung. For the latter, the lower half of the pot was filled with pot soil and the upper half was filled with elephant dung, in which the seeds were placed. Germination and appearance of the first true leaves were monitored at least three times per week for a period of six months. We stopped monitoring 45 days after the last seed in a pot had germinated and no further germination event had occurred.

In addition to the single seed germination experiments, we planted two sets of whole fruits in two subsequent years. In the first year, we half-buried the fruits but recorded no germination success. In the second year we simply placed another 20 fruits on the ground, but unfortunately the experiment was interrupted by heavy rains and flooding before any

germination event might have taken place. We therefore excluded this part of the experiments from all analyses.

4.3.5 Statistical data analysis

As we could not definitively declare the remaining non-germinated seeds as dead, we used survival analysis to calculate the germination rate as a function of time (Lewis 1987; Nchanji & Plumptre 2001; Cochrane 2003; Campos-Arceiz et al. 2008b; Campos-Arceiz & Blake 2011). An important feature of the seed germination data is that the germination times are right-censored due to termination of the experiment before some seeds might still have germinated. For the latter, the exact germination time, thus, remains unknown and they are generally more likely to be censored. As a result, we used the censored and uncensored germination times, with the time in days from planting a seed to the date of germination of the seed as the response variable. We first estimated the distribution function of the seed germination times, i.e., the germination time distribution function (GTDF), and used this function to describe the germination times of the seeds subjected to the different treatments. When evaluated at time *t* the GTDF yields the probability that a given seed from the population of experimental seeds will have a germination time that exceeds *t.* This can be expressed succinctly as

$$G(t) = Pr\ (T > t) \tag{1}$$

where $G(t)$ is the germination distribution function (GTDF) and T is the germination time of a randomly selected seed. We computed nonparametric estimates of the germination distribution function by both the product-limit and life-table methods, also commonly called the Kaplan-Meier and actuarial methods, respectively, in the SAS LIFETEST procedure (SAS Institute 2016). We also computed the closely related function, the cumulative distribution function (CDF):

$$F(t) = 1 - G(t) \tag{2}$$

We further computed the probability density function (PDF) of the germination time, defined as the derivative of $F(t)$, and denoted as $f(t)$ and the hazard function $h(t)$ defined as

$$h(t) = f(t) / G(t). \quad (3)$$

We compared different germination time curves to determine whether the populations of seeds subjected to different treatments had identical GTDF functions. To do this, we used nonparametric *k*-sample tests based on weighted comparisons of the estimated hazard rate of the individual populations under the null and alternative hypotheses, where *k* denotes the number of different treatment groups being compared. We conducted several statistical tests, differing in their weight functions, comprising the log-rank test, Wilcoxon test, Tarone-Ware test, Peto-Peto test, modified Peto-Peto test, and Fleming-Harrington G_ρ family of tests. In the Wilcoxon test for homogeneity, pairs of the germination time functions were compared using the multiple-comparison method and the P-values for the paired tests (raw P-values) adjusted for multiplicity using simulation adjustment (simulated P-values). We performed log-rank and Wilcoxon test, respectively, to test the significance of the association of the germination variable with covariates (category of seeds, dung treatment and planting date of seeds). These tests were conducted by pooling over any defined strata, thereby adjusting for the stratum variables, and were carried out using the SAS LIFEREG procedure (SAS Institute 2016).

If T_i is a random variable denoting the germination time and C_{i1}, D_{i2}, and t_{i3} are covariates denoting the gut passage time category (0,1,2,3), dung treatment (0= without dung, 1= with dung) and planting date (0, 6, 13, 20, 27, 33 days from the start of the experiment) for the i^{th} seed in the sample, then the model for the association between the germination time and the three covariates fitted by the LIFEREG procedure is

$$Log_e(T_i) = \beta_0 + \beta_{i,1,0} C_{i,1,0} + \beta_{i,1,1} C_{i,1,1} + \beta_{i,1,2} C_{i,1,2} + \beta_{i,1,3} C_{i,1,3} + \beta_{i,2,0} D_{i,2,0} + \beta_{i,2,1} D_{i,2,1} + \beta_7 t_{i,3} + \sigma\varepsilon_i \quad (4)$$

where ε_i is a random error term and the βs and σ (scale) are parameters to be estimated. The log transformation of T_i ensures that the predicted values of T are positive regardless of the values of the covariates or their regression coefficients.

4.4 Results

The number of fruits consumed by individual elephants during the feeding trials averaged 15.2 ± 6.2 (n = 76) and ranged between 8 and 25. The mean weight of a single fruit was 427.6 ± 75.4 g (n = 188) whereas the average number of seeds per fruit was 168.9 ± 63.5 (n = 112). A total of 6253 ingested seeds were retrieved from the dung of five elephants over the entire course of the feeding trials. For the sixth elephant we could not ensure a continuous monitoring and therefore excluded this data set from survival rate calculations. The average seed survival rate for five elephants was 79%, based on estimated numbers of seeds per fruit. However, the regression of the number of seeds against the weight of control fruits suggested a weak relationship (r^2 = 0.12, P = 0.000) albeit highly statistically significant and based on an approach used by other studies (Campos-Arceiz et al. 2012; Sekar et al. 2015b). This implies that the reliability of the estimated survival rates of the ingested fruit seeds during their passage through the elephant gut (this should not be confused with the germination rate of the planted experimental seeds) was relatively low. The low reliability arises from the uncertainty associated with the total number of seeds in the ingested fruits estimated from the regression relationship.

Table 4.1. Maximum likelihood parameter estimates for the model relating germination time to the gut passage time (GPT) categories (control, short, medium, long), dung treatment (yes, no) and date of planting seeds. The null hypothesis is that all the coefficients are 0. *exp(β)* is the estimated ratio of the expected (mean) germination times. CL= 95% confidence limits.

Parameter	***Level***	***exp (β)***	***SE***	***Lower CL***	***Upper CL***	χ^2	***P***
Intercept		4.021	0.067	3.891	4.152	3662.7	< 0.0001
GPT	Control	0.588	0.072	0.447	0.729	66.6	< 0.0001
	Short	0.251	0.071	0.112	0.391	12.4	0.0004
	Medium	0.229	0.068	0.095	0.363	11.3	0.0008
	Long	0.000					
Dung	Yes	-0.357	0.045	-0.445	-0.268	62.5	< 0.0001
	No	0.000					
Date		0.025	0.002	0.021	0.029	140.9	< 0.0001
Scale		0.746	0.020	0.708	0.786		

The mean (±1 SD) GPT was 35.3 (± 9.3) h, with a mean minimum of 20 (± 2.1) h and a mean maximum of 72 (± 8.6) h. Of the 1200 seeds planted across all the experimental treatments, 68% germinated and 96% of those that germinated developed first leaves over the course of the seven-month monitoring period. Until the censoring time at 167 days (plus 45 days monitoring without germination event), 61% of control seeds germinated, 69% for short GPT, 67% for medium GPT, and 80% for long GPT, respectively. The mean germination success of seed loads from different elephants was 70% or 105 (± 18.9) seeds per animal. The remaining non-germinated seeds were censored (Fig. 4.1; appendix C.1).

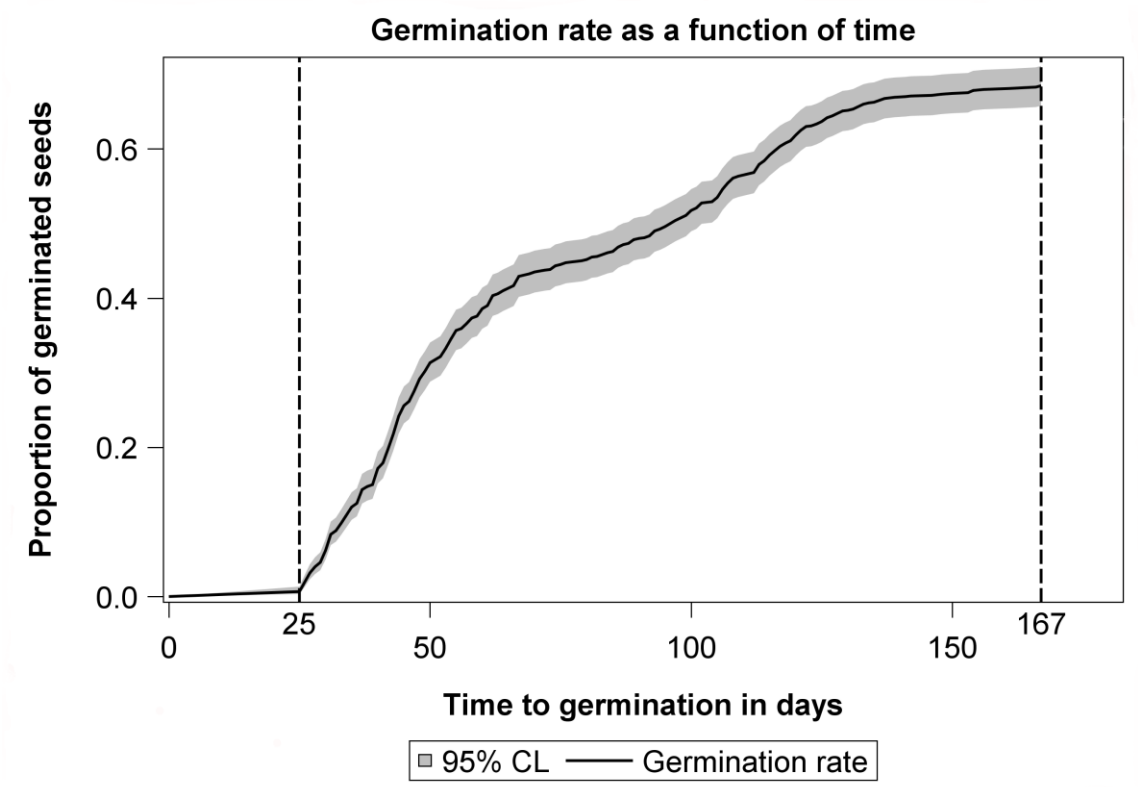

Figure 4.1. Mean proportion and 95% confidence band of germinated seeds across all categories as a function of time to germination. Germination rate =1- proportion of non-germinated seeds. The vertical dashed lines mark the beginning of germination and the right-censoring date, respectively. Monitoring of seeds was continued for 45 days after the date of the last germination event.

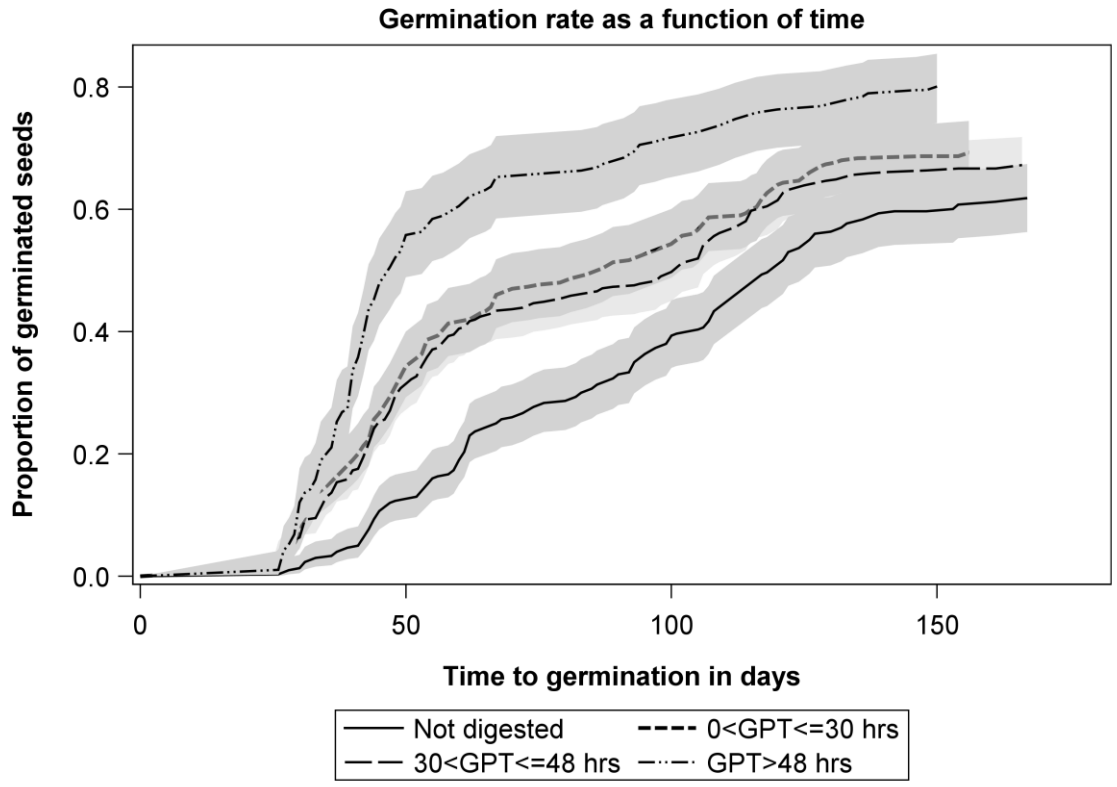

Figure 4.2. The proportion of germinated seeds as a function of time for the control group (not ingested) and the three gut passage time (GPT) treatment groups (short: GPT ≤ 30 h, medium: 30 h < GPT ≤ 48 h, long: GPT > 48 h) and the pointwise 95% confidence bands. Germination rate =1- proportion of non-germinated seeds.

The germination time curves for the four GPT categories (including the control), varied significantly in their expected mean times to germination (Z = 7.77; SE = 24052.94; P < 0.0001; Fig. 4.2, appendix C.2) except for categories 1 and 2 that were similar (χ^2 = 0.4, P = 0.9091, appendix C.3). Germination times were significantly longer for the control (80%), short (29%), and medium (26%) categories, than for the long GPT category (Table 4.1; Fig. 4.2). Similarly, control seeds had longer expected germination times than seeds in the short (Z = 5.24; SE = 0.0642; P < 0.0001), medium (Z = 5.96; SE = 0.0603; P < 0.0001) and long (Z = 8.16; SE = 0.0720; P < 0.0001; Fig. 4.2, appendix C.4) GPT categories. Dung treatment (yes, no) and planting date (0, 6, 13, 20, 27, 33 days from the start of the experiment) had highly significant associations with germination time as shown by the nonparametric Wilcoxon and log-rank tests (P < 0.0001; appendix C.5 and C.6). Results of the LIFEREG procedure of SAS (SAS Institute 2016) provided evidence that GPT (χ^2 = 72.6, P < 0.0001), dung treatment (χ^2 = 62.5, P < 0.0001) and date of planting

(χ^2= 140.9, P < 0.0001) strongly influenced germination time (appendix C.7). The parameter estimates of the regression coefficients showed that the expected germination time is [100 × (1 - $e^{-0.3567}$)] = 30% significantly longer for seeds treated with dung than for the untreated seeds (Table 4.1, Fig. 4.3). The same applies to the median (or any other percentile) time to germination.

The percent increase in the expected germination time for each one unit increase in the planting date is expressed as [100 × ($e^{0.02505}$ - 1)] = 2.54%. This implies that each additional day that passes before the seeds are planted is associated with a 2.54% increase in the expected time to germination, given that the other covariates are held constant. This temporal influence on germination was likely due to the changing climatic conditions over seven months, with longer dry periods in between.

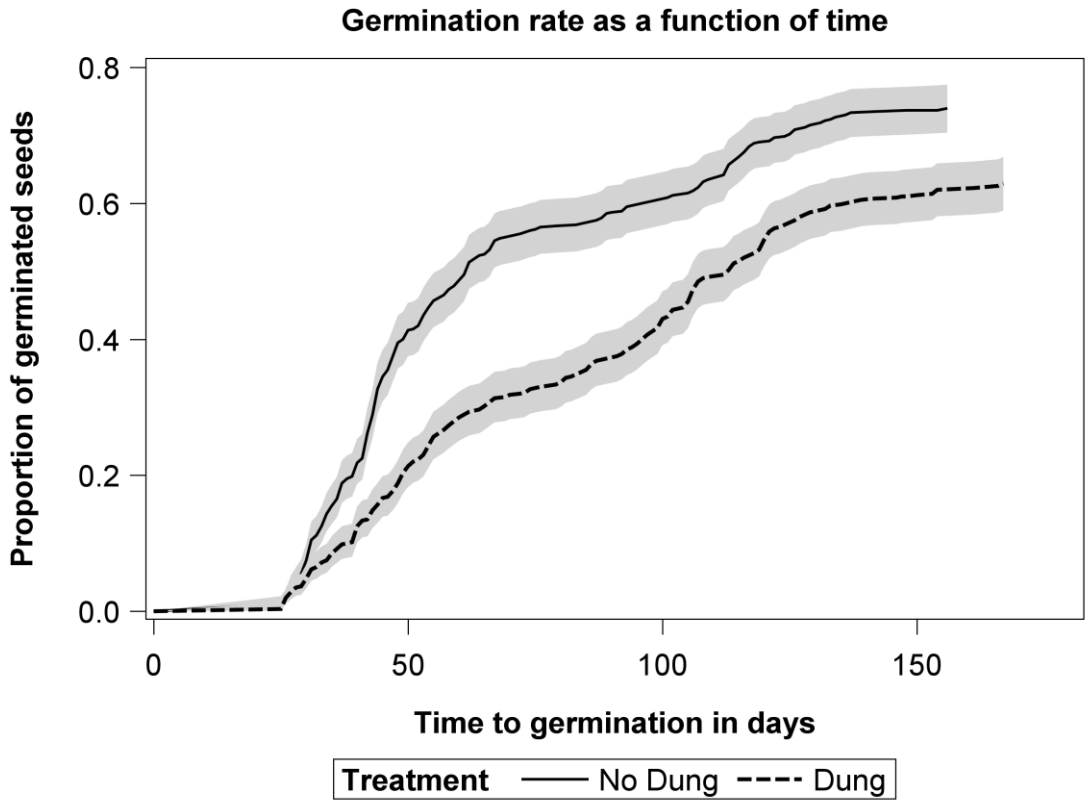

Figure 4.3. The proportion of germinated seeds as a function of time since planting in days for the seeds planted with dung and without dung and the 95% pointwise confidence bands. Germination rate =1- proportion of non-germinated seeds.

4.5 Discussion

4.5.1 Faster germination time for elephant-ingested seeds

In addition to the study of Sekar et al. (2015b) in India, we used a larger sample size of elephants and experimentally evaluated the influence of elephant dung itself on seed germination. Furthermore, we propose and apply a different approach to analyzing germination data by using statistical methods for survival analysis to reduce the potential bias associated with censoring the time to germination of seeds. Our results show that *D. indica* benefits from being eaten, although it does not solely depend on elephants for germination (i.e., a large number of seeds also germinate without being eaten). Surviving post-germination is yet another challenge and the faster germination time for seeds ingested by elephants can be expected to be beneficial if it substantially reduces the risk of seed destruction by post dispersal predators (Allison 1995; Hosmer & Lemeshow 1999). The environmental conditions of the establishing site as well as the type of seed dormancy additionally influence germination speed (Schupp 1993; Traveset & Verdú 2002; Cochrane 2003). Elephant dung has so far been found to provide neutral or beneficial environmental conditions in the form of nutrients, humidity and protection from predation (Crawley 2000; Traveset & Verdú 2002). Surprisingly, we found that *D. indica* seeds planted with dung had a longer germination time than seeds planted without dung, which might have been due to the limited pot size and no interaction with surrounding soils and fauna. In natural conditions, the intraspecific competition of large amounts of seeds deposited in the same dung pile might reduce seedling success (Campos-Arceiz & Blake 2011). The seed load naturally depends on how many fruits the elephants consumed, which, in our study differed across individuals, with 25 fruits being the maximum amount eaten. Sekar et al. (2015b) observed individual differences across elephants, ranging from 7 to 52 fruits of *D. indica* being eaten. Generally, for elephants in natural habitats it is well documented that *D. indica* is a welcome addition to their usual diet (Lewis 1987; Campos-Arceiz & Blake 2011), and the elephants in our study seem to conform with this observation.

4.5.2 Germination success increases with gut passage time

We found that *D. indica* seeds that had the longest gut passage time had the highest germination success. The gut passage can have positive, negative or neutral effects on seed viability (Campos-Arceiz et al. 2008a; Datta & Rawat 2008; Sekar & Sukumar 2013). In

our study, ingested *D. indica* seeds, regardless of their GPT category, had a higher germination rate compared to non-ingested control seeds. One challenge in seed germination experiments is that observation time is often limited and potential later germination events might be missed. Several studies have addressed this challenge by testing whether the remaining non-germinated seeds contained a viable or a rotten embryo (e.g.Campos-Arceiz & Blake 2011) and/or by continuing monitoring until a certain time after the last seed has germinated (Campos-Arceiz et al. 2012; Sekar et al. 2015b; our study). The risk of bias remains due to the potential censoring of data and wrongly pronouncing potentially viable seeds as non-viable. We used survival analysis to minimize both potential sources of bias (Chapman et al. 1992; Nchanji & Plumptre 2003; Campos-Arceiz et al. 2008b). The proportion of germinated seeds as a function of time allowed the comparison of germination success among the different categories at any given point in time until the censoring date and provided information on the category-specific speed of germination.

4.5.3 Large dispersal distances through elephants

Apart from the faster germination of ingested seeds, another benefit for *D. indica* from the seeds consumed by elephants might be the seed dispersal distances and their impacts on the seed shadow (the distribution of viable seeds around their source; (Allison 1995). Asian elephants have home ranges of 50 - 1000 km^2, reflecting the large area across which they can alter or maintain plant composition in ecosystems (Janzen 1971; Willson & Traveset 2000). The seed dispersal distance by elephants varies with the size of the plant seed consumed, with larger seeds taking more time to pass through the digestive system (Sukumar 1989; Sukumar 2006; Campos-Arceiz et al. 2008b). *Dillenia indica* seeds (~6 mm in size) are relatively small and, hence, remain in the digestive tract for a rather short time period; their maximum GPT of 72 h we found is much shorter than the maximum GPT of 114 h, reported for tamarind seeds (*T. indica*), which are about twice as large in size (Campos-Arceiz et al. 2008b). Mean dispersal distances for the latter were found to be about 1-2 km in Myanmar and Sri Lanka, depending on the season (Campos-Arceiz et al. 2008b), while Sekar et al. (Powell 1997) recorded mean dispersal distances of about 3.5 km for *D. indica*, *Artocarpus chaplasha*, and *Careya arborea* in India. Tamarind seeds were negatively affected by the retention time in the gut (Sekar et al. 2015b). In contrast, *D. indica* seeds in our study profited: the longest GPTs and, therefore, the largest potential

dispersal distance had the highest germination success. Hence, our findings highlight *D. indica´*s high adaption to and potential benefit from megafaunal dispersers.

4.5.4 Other potential means of seed dispersal

With decreasing numbers of elephants and other megaherbivores as seed dispersers, plants have to rely on alternative means of dispersal such as livestock, humans or water, which has been reported for *D. indica* (Van Steenis 1948; Donatti et al. 2007; Datta and Rawat 2008; Campos-Arceiz et al. 2008b). This might not apply for areas with longer dry periods, where smaller animals might contribute to seed dispersal (Sekar and Sukumar 2013). Elephants remove significantly more fruits than other animals such as bovids, macaques and rodents but all of these species were able to access the seeds once the mesocarp had softened (Sekar and Sukumar 2013). While removal does not necessarily lead to dispersal, rodents, for example, are known to store seeds, thereby sometimes contributing to dispersal (Donatti et al. 2007). Also macaques can serve as effective seed dispersers but in several cases have negative impacts on germination and viability of some species depending on the temporal context (Forget et al. 2002; Hulme 2002; Vander Wall 2002). Domestic bovids, on the other hand, are able to disperse a great number of seeds for some species as well, but do not reach the seed dispersal capacity of elephants (Albert et al. 2013; Tsuji 2014).

4.5.5 Conclusion and conservation implications

In times of climate change, a large dispersal area might become increasingly important as it might help in buffering off potential population losses due to adverse environmental conditions (Sekar et al. 2015b). However, due to increasingly intensive land use and destruction of ecologically important forests, movements of large mammals are becoming increasingly impeded with the result that their seed dispersal potential might either rapidly decline or even disappear altogether. Large-seeded plant species in particular are at a greater risk of being negatively affected by selective logging and hunting as large seed-dispersing frugivores are often the first animals to vanish from disturbed forests (Markl et al. 2012). Plant species experiencing the loss of their main seed dispersing animal agents might suffer collapses in their recruitment and regeneration cycles (Guimarães Jr. et al. 2008; Blake et al. 2009). Likely consequences will be increased clustering of tree populations and lower dispersal distances with associated reductions in the overall

geographic range as well as losses in genetic variation (Corlett & Westcott 2013). Changes in species composition are to be expected with particularly severe ecological shrinkage in isolated ecosystems (Cramer et al. 2007; Terborgh et al. 2008; Markl et al. 2012).

Our results show that *D. indica* does not solely depend on but seems to benefit from being eaten by elephants as ingested seeds were significantly more likely to germinate and to do so earlier than non-ingested control seeds. With this study we contribute to the understanding of the effects of Asian elephants' frugivory which has been much less researched than that of African ones (Pérez-Méndez et al. 2015). While we still know relatively little about elephant seed dispersal, particularly in Asia, it is clear that elephants hold key functions in forest ecosystems. The megaherbivores shape ecosystems through their high food intake and by destroying vegetation through trampling or breaking, thereby acting as filters on tree recruitment and shifting balances of herbaceous and woody plants (Hansen & Galetti 2009; Campos-Arceiz & Blake 2011). They are likely helping to avert exceeding redundancy while maintaining plant diversity and thus further decline or local loss of elephants and other large herbivores would likely favor abiotically-dispersed species, leading to simpler plant communities (Bakker et al. 2016). The disappearance of elephants could further trigger cascading effects for overall system functioning through alterations in habitat and trophic structures, leading to changes in abundance or even extinction of other animal species down to potential deterioration of carbon storage and disturbances of nutrient cycles (Malhi et al. 2016; Bello et al. 2015; Terborgh et al. 2016). Hence, their stringent protection will not only benefit the pachyderms themselves but also aid in conserving the habitat for a broad range of plant and other animal species, and ultimately sustaining the services such forests provide also for humankind.

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Chapter 5

Biodiversity and Ecosystem Services - A case study for the Assessment of Multiple Species and Functional Diversity Levels in a Cultural Landscape[4]

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5.1 Abstract

The expansion of large-scale plantations has a major impact on landscapes in the Tropics and Subtropics. Crops like soy bean, oil palm and rubber have led to drastic changes in land cover over the past decades, thereby altering ecosystem functions and services (ESS). Associated shifts in ESS such as climate regulation, erosion and water cycles, biodiversity as well as soil fertility or the provisioning of raw materials have been assessed through several models and software solutions (InVEST, ARIES, MIMES). However, suitable methods for the integration of a range of biodiversity assessments in agricultural landscapes are scarce. With this study, we introduce a methodology for incorporating multiple levels of species diversity into models to allow an integrated evaluation of ESS. We collected data sets from both published and unpublished sources on the distribution of vascular plants, selected pollinator groups, ground beetles, ungulates as well as amphibians, mammals, reptiles and birds in rubber-dominated landscapes, with a focus on our study sites in Southwest China and Thailand. Based on this information, we developed a common classification scheme that enables the integration of different facets of biodiversity (species diversity and functional diversity) to complement an interdisciplinary ESS assessment. Species diversity data were normalized against the most divers habitats

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reported (using habitat scores) to assess the impact of rubber cultivation on multiple levels of biodiversity. This resulted in a comparable matrix of different land use types and their suitability as habitat for the respective species groups allowing the aggregation of very diverse indicators. The findings were applied to two alternative land use scenarios in southern China to highlight the potential effects of land use and management decisions on species and functional diversity. Our results highlighted that the conservation oriented scenario did score higher for habitat suitability in both total species (+5%) as well as IUCN Red List species (+6%) assessments compared to the current state or business as usual scenarios (-2% and -3% compared to current state). The process presented here allows for an application within established ESS software programs, in our case InVEST, using aggregated indices while additionally providing enhanced opportunities for comparable, spatially explicit assessments of the expected impact of the analyzed scenarios on specific species groups.

Key words: *Habitat suitability, indicator development, rubber, ecosystem service assessment, InVEST, biodiversity*

5.2 Introduction

Although biodiversity is not an Ecosystem Service (ESS) in the narrow sense, it is inextricably linked to ESS because biodiversity influences their provision (MEA 2005). Several analytical tools exist to allow for replicable and quantifiable ESS analyses, such as InVEST (Integrated Valuation of Ecosystem Services and Trade-offs), ARIES (Artificial Intelligence for Ecosystem Services, 2016a, b; Villa et al. 2014) and MIMES (Multiscale Integrated Model of Ecosystem Services), which all belong to the group of independently applicable and generalizable landscape-scale models (Bagstad et al. 2013a). InVEST is provided by the Natural Capital Project (The Natural Capital Project 2015; 2016) and has a modular approach. Its tools are based on deterministic production functions; two of them cover biodiversity issues: habitat quality and habitat risk assessment. The habitat model of InVEST has already been used in different scientific analyses (e.g. Dhakal et al. 2014; Terrado et al. 2016). ARIES includes not only the provisioning of ESS and possible trade-offs, but also the flow and the use of ESS. The current release includes eight ESS and even though there is no explicit model on biodiversity, the link between biodiversity and ESS is stated frequently. Although InVEST and ARIES differ greatly in their methodology, a

comparison of the two tools yielded similar results for the ESS carbon, water and scenic view sheds for a case study in Arizona (Bagstad et al. 2013b). MIMES aims at supporting sustainable and ecosystem-based management planning with a special focus on incorporating concepts to capture the dynamic character of coupled human and natural systems. Biodiversity is included in the modeling process as being part of the biosphere, but neither explicitly mentioned as output nor included in the research questions so far presented in the literature (Boumans et al. 2015).

Out of these three possibilities we chose InVEST for our analyses because it has the most detailed routine for the evaluation of biodiversity within the framework of available ESS assessment tools. By combining aspects of habitat suitability with the potential to analyze alternative scenarios of land use decision, it was the most suitable tool to meet our research questions. In addition it is open-source, well-documented and the models have a high generalizability (Bagstad et al. 2013a).

Human activities have a strong impact on the distribution of habitats, land cover and landscape patterns throughout the world. Deforestation, transformation, degradation and intensification impact the quality and quantity of ESS provided by a given landscape; the potential as habitat for species being one of these (Ahrends et al. 2015). The expansion of rubber (*Hevea brasiliensis)* in mainland Southeast Asia (MSEA) is providing a suitable case study to asses some of these impacts (Ziegler et al. 2009), as a range of studies have covered the implications for single species groups, but so far a general assessment is amiss. The shift from low input agriculture to intensified systems additionally reduces functional and species diversity and consecutively, the availability of potentially useful habitat types for species originating from natural forest. Rubber plantations generally harbor less than half of the original fauna and flora species richness compared to natural forest. Most studies therefore found that many of these species are unable to exist permanently in rubber plantations (Beukema & van Noordwijk 2004). There is clear evidence that the existence of natural forest area is essential for the conservation of large proportions of native forest species in rubber-dominated landscapes (He & Martin 2015).

5.2.1 Study sites

The research area to which our model is applied is the Naban River Watershed National Nature Reserve (NRWNNR, 22°08'N 100°41'E) in Xishuangbanna, Yunnan Province, PR China. The nature reserve covers 271 km² and its elevation ranges from 500 m to 2300m a.s.l., covering the watershed of the Naban River, a tributary of the Mekong River. It features an especially high diversity of natural vegetation types, as well as a variety of land use systems due to the topographically and ethnically diverse background of the region (Zhu et al. 2005). In Xishuangbanna, economic development and biodiversity conservation compete for the same land. So far, economic goals have dominated: from 1976 to 2003, 140,000 ha of tropical rainforest were replaced by rubber (Li et al. 2005). Since then, forest clearing has continued.

Data on mammal diversity were mainly collected in the Province Surat Thani in southern Thailand, in and around the Tai Rom Yen National Park (TRY). TRY is located in the East of the province and covers an area of about 400 km^2 with an elevation range from around 100 to 1200m a.s.l.. The protected area was created in 1991 and includes evergreen forest, partly characterized by limestone formations (DNP 2013a), as well as cultural landscapes dominated by rubber plantations. This region is a traditional rubber cultivating area (Li & Fox 2012) that lies within the transition zone of the Indo-Burma and Sundaland biodiversity hotspots, home to more than 300 mammal species (Myers et al. 2000).

5.3 Material and Methods

5.3.1 General Approach

With this study we aim to introduce a methodology for incorporating multiple levels of species diversity into models to supplement the evaluation of ESS. We have combined original data from field work within our study sites with complementary data on biodiversity from detailed literature review studies. In order to make these results comparable we conducted a normalization process and integrated this data into the spatially explicit model habitat suitability model of InVEST.

5.3.2 Own studies on site

Floristic inventory data were analyzed considering the occurrence of rare and endemic species according to the IUCN (International Union for Conservation of Nature) red list of species, and the value of a vegetation type for human use, expressed through the number of medicinally usable plants. A total of 18,901 m^2 of land in NRWNNR were surveyed based on 610 plots, with different sizes (1m² for rice paddies to 400m² for forest plots) for different land use types (see Table 5.1). Some 1,252 species from 635 genera and 158 families were identified. More details (sampling structure, size bias and categorization) are described in Cotter et al. (2014).

Table 5.1. Distribution of different land use classes between scenarios. Values are given in %.

%	Forest	Rubber	Rice	Agriculture	Bushland
Land Use 2007	69.5	9.5	3.0	15.3	2.2
Go green 2025	78.0	6.3	3.3	7.5	4.8
Business as usual 2025	67.6	9.6	3.3	15.2	3.1

We assessed the presence of ungulates using camera traps, transect surveys and spoor plots in two different protection zones in 2010, for detailed data collection see (Treydte et al. 2013). Mammal diversity data were collected in the years 2013 and 2014 through camera traps, transect surveys and interviews with local farmers living in the periphery of the TRY National Park in Thailand. In Thailand, assessments were undertaken inside the natural forest, in the adjacent farmland and at the forest-farmland boundary. A total of 180 randomly selected farmers were interviewed around the TRY National Park about wildlife species and number of individuals that came to the farmland. Camera traps were repeatedly installed at 30 locations along the forest edge around the protected area. Further, 21 locations were sampled several km into the forest, along wildlife trails. Cameras remained at least two weeks in one location. In addition, data were collected on 25 one-km trail-transects within the forest and along 25 complementary transects up to one km into the farmland, in perpendicular distance to the forest border. All mammal species were

considered but for small species such as mice, rats or squirrels identification to species level was often not feasible and most animals recorded were assigned to the overall taxonomic group of 'rodents', resulting in a non-proportional low species number for this group. Other groups for which we recorded sufficient species were ungulates, carnivores and primates. Various small species with a weight of less than 7 kg belonging to different taxonomic groups were compiled in the group 'other small mammals', e.g. treeshrews (*Tupaia glis*), pangolins (*Manis javanica*) and colugos (*Galeopterus variegates*).

5.3.3 Literature research

By using Scopus bibliographic database, we investigated published articles dealing with animal biodiversity in rubber plantations in comparison to natural habitats, and other land use types when indicated. For data analysis, we selected studies for two criteria: (a) precise and comparative information on species numbers in rubber plantations and natural habitats, (b) studies located in MSEA, assuming that the ecological conditions and land use types in this region are largely comparable to the situation in the target area of NRWNNR. Therefore, we did not include studies from other regions such as India and Indonesia, as well as studies conducted in rubber agroforestry systems ("jungle rubber"). Accordingly, we considered a total of 10 studies, including soil nematodes, different taxa of insects, spiders, frogs, bats and birds.

5.3.4 Data integration

The data sets from field and literature studies were subjected to a normalization process aiming at transforming species (group) numbers to comparable and computable values, named habitat scores, between 0 and 1. The highest reported species number was used as reference value (1), while the species numbers reported for other land use classes were transformed into a fraction of the reference value (e.g. plant species numbers; see Table 5.2: Forest 796, Rubber 518, rice paddies 146 resulted in normalized species indices of Forest 1, Rubber 0.65, Rice paddies 0.18.). These indices were aggregated into a habitat suitability matrix that served as input into our InVEST scenario evaluation. Reference values for the aspect of habitat sensitivity to different threats needed for an InVEST assessment were derived from literature review and decided upon during expert panel meetings.

We analyzed (1) red list species (flora and mammals) for the biodiversity aspect of nature conservation (red list species), (2) an overall biodiversity score including all species for an overall picture (all species) and (3) the combination of bees (honey provision) and medicinal plants to show from which species humans benefit directly (human use). For all groups spatially explicit results were modeled with the habitat quality model of InVEST 3.2.0 for the land use map of 2007 and two scenarios for 2025.

5.3.5 Case study scenarios

The original land use map for our study site in Southwest China was created from 2007 data (Wehner et al. 2014). In order to test our approach we applied our methodology to two scenarios for the year 2025, the 'Business as usual' scenario extrapolating the trend of further expanding rubber plantations and the 'Go green' scenario increasing the available forest area. These scenarios have been developed to evaluate current land use composition and to establish possible future trajectories of land use change in the region (Wehner et al. 2014) using CLUE-S modeling approaches combined with socio-economic modeling techniques (Gibreel et al. 2014) and combined ecological assessment and economic optimization approaches (Cotter et al. 2014).

We assessed habitat suitability based on our collected data sets with InVest in order to assess the impact the different land use scenarios would have on biodiversity as an ESS. Further studies will report on the integration of this assessment into our integrated multidisciplinary assessment approach.

5.4 Results

All groups of vertebrates and invertebrates listed in Table 5.2 showed lower species numbers in mature rubber plantations compared to natural forest, latter representing the original type of vegetation in MSEA. The proportional abundance of species in these two land use categories varied widely according to taxa and study. In most cases, species numbers were roughly 30-50% lower in rubber plantation compared to forest.

The results showed that hoverflies occurred in much higher species numbers in agriculture and rice fallows than in forest and rubber plantations (Meng et al. 2012b). Species numbers of ground beetles in forest and agricultural land use classes were nearly identical but much

lower in rubber plantations (Meng et al. 2012a). On the other hand, higher species numbers of longhorn beetles and birds were found in forest compared to rice and agriculture (Meng et al. 2013; Peh et al. 2006). Generally, relatively high numbers of species in agricultural habitats can be explained by the occurrence of additional species that are adapted to open land conditions and do not originate from forest.

Small vertebrates such as frogs, birds and rodents (Aratrakorn et al. 2006; Peh et al. 2006; Behm et al. 2013; Li et al. 2013) showed a lower diversity in rubber-dominated land use systems than in natural forests but generally occurred in higher numbers than larger vertebrates such as carnivores or ungulates across all land use classes. The latter rely on natural forest for shelter and food, whereas the reduced cover in plantations makes them easy targets for hunters (Luskin et al. 2014). The diversity and activity of insectivorous bats is greatly reduced in rubber plantations because insects are less than half as abundant as in natural forest habitats (Phommexay et al. 2011).

We used the results of Table 5.2 to model habitat suitability scores for three different groups of species within the NRWNNR. Information on the input data for the model is available in the appendix. The color bar shows increasing habitat suitability from zero to one (red to dark green, respectively). For the 2007 land use map and both scenarios habitat suitability is worst for red list species and best for species used by humans, with in-between scores for the aggregated biodiversity index of all species (Fig. 5.1). When comparing the two scenarios with the land use map of 2007 within species groups the ‘Business as usual’ scenario slightly decreases habitat suitability, whereas the ‘Go green’ scenario slightly increases habitat suitability. Landscape habitat scores (sum of all pixel habitat scores) for the ‘Business as usual’ scenario vary between 97-99%, for the ‘Go green’ scenario between 103 and 106%. Again, the magnitude decreases from red list species through all species to human use species. We also created a benchmark scenario to assess how habitat suitability would behave over a longer time span by converting all land use classes except settlements to forest. Using the resulting landscape habitat score as 100% benchmark the landscape habitat scores for land use map of 2007 is 79%, 83% and 90% for red list species, all species and human use species respectively.

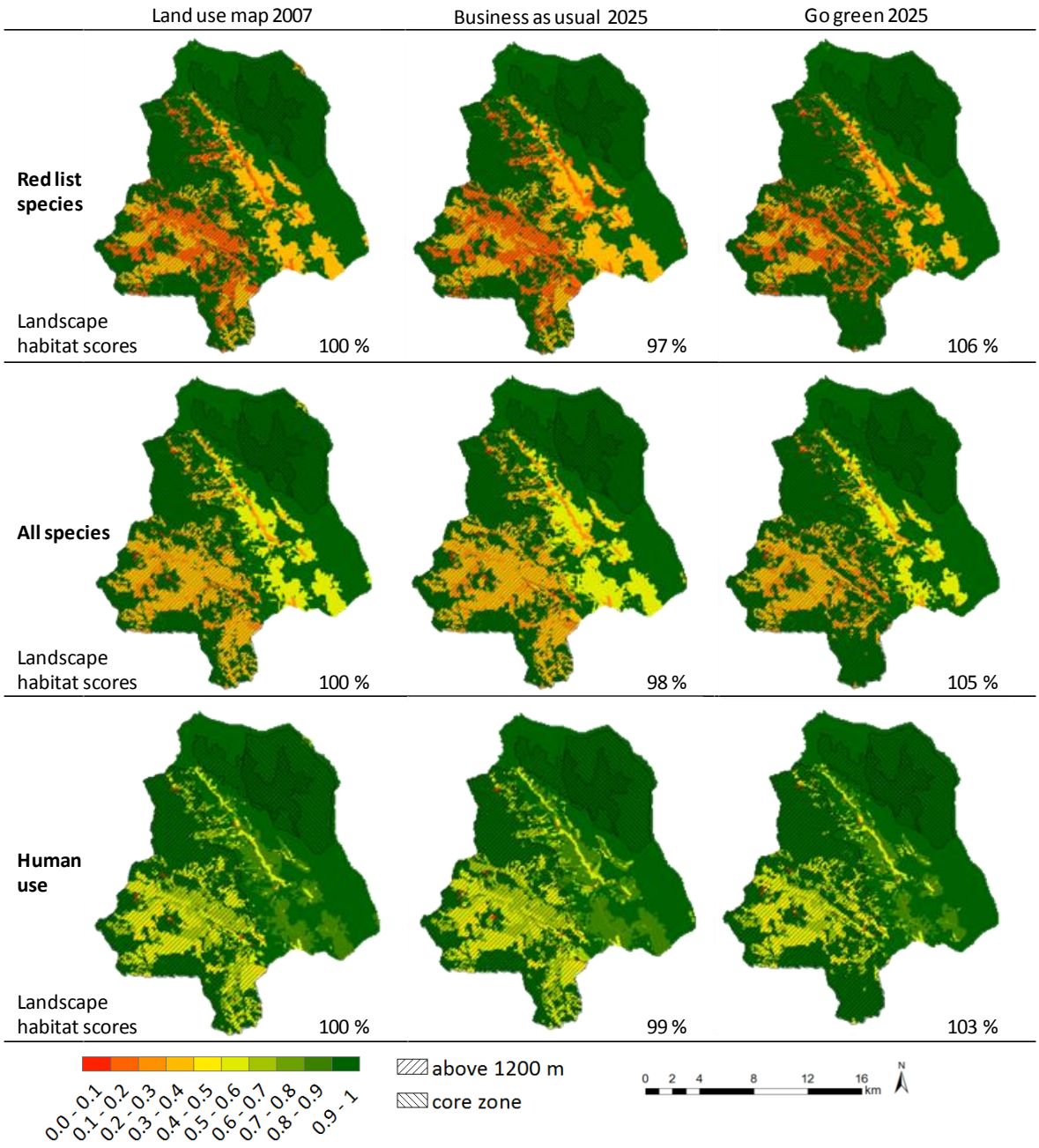

Figure 5.1. Displays the modeled habitat scores for the land use map of the year 2007 and the two scenarios for 2025, 'Business as usual' and 'Go green'. Values close to zero designate areas with no or a low habitat suitability for the respective species groups (red list species, all species and species with a human use value), values close to one designate areas with a high habitat suitability (forest was used as reference for highest habitat suitability). Hatched areas cannot be converted to rubber plantations either because they are above 1200m a.s.l. or legally protected (core zone of the Man and Biosphere Programme).

Table 5.2. Results of species numbers for flora, vertebrates and invertebrates per land use class (absolute numbers and proportions). Habitat scores are calculated as the proportion of species found in a specific land use class related to a benchmark value, which is the land use class with the highest absolute number of species found (100%). Studies from different countries are listed separately (e.g. ungulates, birds).

	Forest		Rubber		Rice		Agriculture		Bushland		
	n	*%*	n	*%*	n	*%*	n	*%*	n	*%*	sources
Flora											
Red list	91	*100*	20	*22*	1	*1*	6	*7*	3	*3*	Own, 2009
Endemic	126	*100*	45	*36*	2	*2*	15	*12*	3	*2*	Own, 2009
Invasive	7	*35*	20	*100*	20	*100*	18	*90*	14	*70*	Own, 2009
Medicinal	209	*100*	193	*92*	60	*29*	92	*44*	55	*26*	Own, 2009
Total species	796	*100*	518	*65*	146	*18*	255	*32*	124	*16*	Own, 2009
Habitat scores[1,2]	**1**		**0.65**		**0.18**		**0.32**		**0.16**		
Vertebrates											
Red List mammals	10	*100*	5	*50*							Own, Thailand, 2013
Ungulates	6	*100*	2	*33*							Own, Thailand, 2013
Ungulates	6	*100*	3	*50*							Own, China, 2010
Primates	6	*100*	3	*50*							Own, Thailand, 2013
Carnivores	10	*100*	3	*30*							Own, Thailand, 2013
Rodents	7	*100*	5	*71*							Own, Thailand, 2013
Mammals < 7 kg	5	*100*	3	*60*							Own, Thailand, 2013
Bats (insect.)	26	*100*	13	*50*							Phommexay et al. 2011
Frogs	18	*100*	11	*61*							Behm et al. 2013
Birds	108	*100*	41	*38*							Aratrakorn et al. 2006
Birds	53	*100*	43	*81*							Li et al. 2013
Birds	62	*100*	46	*74*					31	*50*	Peh et al. 2006
Habitat scores[1,3]	**1**		**0.56**								
Invertebrates											
Wild bee	35	*100*	25	*71*	26	*74*	30	*86*			Meng et al. 2012b
Hoverflies	12	*36*	2	*6*	33	*100*	31	*94*			Meng et al. 2012b
Ground	59	*92*	34	*53*	64	*100*	53	*83*			Meng et al. 2012a

beetles									
Longhorn beetles	193	*100*	77	*40*	33	*17*	82	*42*	Meng et al. 2012a
Soil Nematode taxa	42	*100*	28	*67*					Xiao et al. 2014
Spiders	193	*100*	95	*49*					Zheng et al. 2015, averages
Habitat scores[1,4]	**1**		**0.49**		**0.29**		**0.37**		
All Species									
Habitat scores[1,5]	**1**		**0.57**		**0.16**		**0.23**		

[1] Habitat scores are calculated as the proportion of species found in a specific land use class related to a benchmark value, which is the land use class with the highest absolute number of species found (100%).

[2] Based on total species found. The benchmark for flora is for all plant groups forest, except for invasive species (highest species numbers found in rubber and rice).

[3] Based on the sum of the listed species groups. The benchmark for vertebrates is for all groups forest.

[4] Based on the sum of the listed species groups. The benchmark for invertebrates is for all groups forest, except for hoverflies (highest species numbers found in rice).

[5] Mean values of overall habitat scores of flora, vertebrates and invertebrates.

5.5 Discussion and Conclusions

With this case study, we were able to demonstrate that our approach of combining multi-species data supplemented with literature data (allowing for a broader assessment of the impacts of land use decisions on biodiversity) can be used as input into established ESS assessment frameworks. The habitat suitability for the species groups studied has changed over the course of the scenarios as expected. Due to the relatively narrow (or realistic) definitions of the scenarios (e.g. rubber area +15% and -30%, continuously strong dominance of natural forests), these changes have been less pronounced as we had expected. Especially when considering the indices from the category "human use" we see less of an impact between the scenarios. This can be explained by looking at the distribution of these species groups within the land use classes (Table 5.2). Medicinal plants for example are still widely present within rubber plantations, albeit with different species (generalists, ruderal flora). A shift between rubber and forest thus has a relatively

low impact on the number of species with medicinal use at least for the species considered. In addition, wild bee species and ground beetle species are present both with forest dwelling species and species adapted to open-land or succession stages found in young rubber. This points towards one of the weaknesses of this method: the decision-making on "what" to analyze and "how" to group species. The negative effects on habitat suitability, especially when considering plant diversity, might prove to be considerable more pronounced when including studies on "second generation" rubber plantations, where rubber has been the dominant land use pattern for more than 30 years.

When taking a look at total species numbers within land use classes, the forests in our study site are ranking top. Considering the landscape level, we expect a mosaic of forests with high diversity of endemics combined with a variety of agricultural activities to still offer a reasonably high number of habitats and thereby support a diverse, but differently composed species community due to the newly introduced farmland species. The method used in this case study allows for the inclusion of sub-assessments such as "Red List species" or "endemics" into the evaluation in order to cover not just quantity, but also "quality" and "characteristics" of species diversity.

For this study we hypothesized that rubber management practices throughout MSEA have similar impacts on biodiversity. We wanted to show a potential approach for combining multiple sources of data in order to allow for a broader assessment of the potential impacts that certain decisions may have on ESF and ESS. To do so, we had to make compromises on the level of details included, a situation that most scientists face in data-scarce environments. While rubber served as a good example to test our approach, management practices within the plantations might have had an impact on habitat quality that we could not cover within our case study. Another aspect to be considered is the difficulty to include different succession stages in rubber plantations. Over the cultivation period the rubber system offers habitat for multiple and successive communities of species, from open land and forest remnant species in the establishment phases up to (moderately) shade tolerant species usually found in secondary forest systems. In the four insect studies for example it was stated that young rubber sites generally harbor higher numbers of insect species than mature rubber plantations due to more comprehensive spatial heterogeneity and the existence of pronounced ground cover vegetation. Mature rubber plantations in contrast are

characterized by lower heterogeneity and their intensity of production which negatively affected food availability for several larger wildlife species.

The method presented here highlights how the predictive power of ecological indicators such as habitat suitability can be improved in data scarce environments. We successfully combined genuine research data from local field studies including different taxa with an extraction of applicable data on species diversity taken from a review of representative studies. Our interdisciplinary work combined data from different fields of ecological research (Fauna, Flora, Landscape) across various spatial scales. In contrast to other publications using the InVEST habitat quality model we included a variety of species in combination with habitat scores that are regionally adapted (Baral et al. 2013; Alarcon et al. 2015; Terrado 2016). We further calculated overall habitat scores based on multiple genus level data sets and derived group level threat sensitivity values while for example Dakhal et al. (2014) only used threats on one single habitat suitability index. With this novel approach we allow for a broader and more representative application of the indicator without having to shift the latter's explanatory power into the discipline of the underlying field studies.

The method presented in this paper is quite robust to scale effects. By including the normalization process we aimed at analyzing the effects of species establishment and integration into communities within a given landscape matrix rather than assessing specific species diversity alone. As such, data from rather small scale assessments can be compared to those from larger ones. At the same time this approach can easily be applied to a wide range of study sites, as the techniques used are not restricted to tropical environments or certain agricultural practices. Our current work is focused on the Greater Mekong Subregion, but the underlying concept can easily be transferred to other regions that face similar problems concerning land use change. Situations of special interest could be the establishment of plantations for the production of renewable primary products such as oil palm in South East Asia, the large scale production of animal feed in the Amazon basin and Cerrado of Brazil, or the changes in landscape composition caused by smallholder farmers on the margins of the rainforests of Africa and South America. Tools such as InVEST have shown to be usable across multiple spatial levels and, when accompanied by data gathering approaches similar to the one presented in this case study, can be a valuable

tool to derive meaningful indicators for ecological impact assessment in land use planning and scenario assessment.

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Chapter 6 General Discussion

6.1 Overview

This thesis focused on conflicts of rubber cultivation and wildlife emerging from conservation areas with a special focus on the endangered Asian elephant. As a consequence of the expanding human land-use in Southeast Asia and decreasing natural habitat, conflict levels are likely to increase in the future, thereby threatening both rural farmers' livelihoods as well as the survival of wild mammal populations. The findings of this study can serve as an essential basis for the development of measures to consolidate farming and conservation interests.

The wild mammal species diversity in rubber plantations was found to be generally much reduced compared to natural forest, particularly with increasing distance from latter habitat (chapter 2 & 3). Most species found in the cultivated landscapes were likely visitors rather than residents. This highlights the importance of softening the forest-farmland edge, e.g., by retaining higher levels of natural undergrowth in rubber plantations. Elephants were responsible for the vast majority of wildlife damage incidents in rubber but most of these cases affected young rubber plants that were not yet tapped (chapter 3). Also, less than half of the farmers who have had elephants on their farms suffered crop damages. These findings indicate that rubber-dominated landscapes surrounding protected areas have the potential to facilitate coexistence between people and elephants if young plants are better protected and plantation management is made more wildlife-friendly. This is of importance as wildlife and particularly elephants do not only inflict costs but also deliver ecological benefits from which ultimately people can profit as well. The study in chapter 4 investigated the ecological function of elephants as seed dispersers and the findings indicate that the exemplary tree species *Dillenia indica* benefits from dispersal by the mega-herbivores. Their declining numbers might, therefore, have long-term negative consequences for certain tree species and overall forest biodiversity and structure.

Biodiversity including mammalian wildlife diversity is an integral and essential component of ecosystem functioning and the provisioning of services to humankind. Hence, chapter 5 presented a methodology which we developed to include multiple levels of species diversity into an integrated ESS evaluation model. The land-use scenario with a

conservation focus, e.g. enrichment of plantations and increasing of natural forest cover, scored the highest habitat suitability for all species levels, thereby further highlighting the importance of protecting natural wildlife habitat for overall biodiversity as a foundation for ecosystem functions and services.

This overall discussion addresses the methodological approaches, combinations and limitations of the study. The costs and benefits of wildlife conservation in rubber-dominated landscapes are discussed in terms of the socioeconomic importance for the farmers' livelihoods as well as the ecological functions and services provided by wild mammals. Finally, future steps for more sustainable and conflict-reduced rubber cultivation practices are proposed.

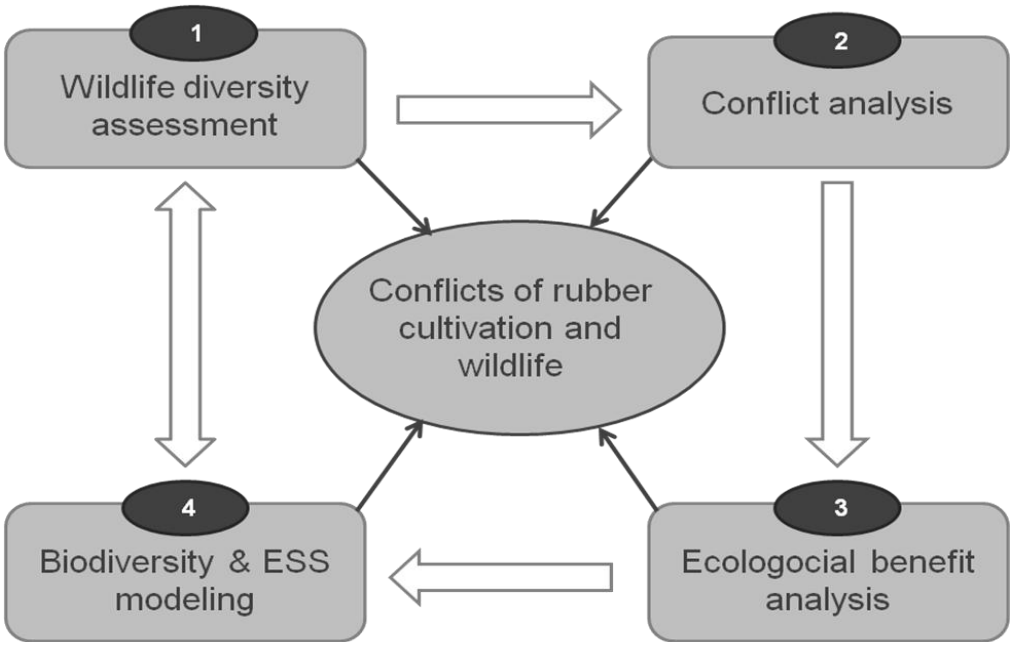

Figure 6.1. Overview of methodologies and targets of this thesis.

6.2 The Elephant in the Dark

6.2.1 Comprising multiple sources of data collections for a comprehensive picture

The parable of the blind men and an elephant tells the story of a group of blind men who touch an elephant to find out what it is. As each of them investigates a different part of the animal, they come up with varying ideas such as a pillar (for the foot) or a brush (for the

tail). This old Asian tale describes vividly the difficulties in obtaining an accurate picture from a single perspective and from a snapshot of a situation. Similarly, accounting for differences due to methodological approaches is particularly important in scientific studies on highly complex and variable ecological processes (Boitani & Fuller 2000). Effective conservation strategies require a profound knowledge on the management needs and challenges, including ecological, socio-economic, cultural and political driving factors of threats (Sutherland et al. 2004; Meijaard et al. 2011). Therefore, multiple assessment approaches were applied in the framework of this thesis to analyze the wild mammalian species richness, presence and distribution in rubber-dominated areas as well as costs and benefits of conserving large mammals such as the Asian elephant in these landscapes.

The knowledge of local people on the kind and occurrence of wildlife species represents an important data source in ecological assessments, particularly in the case of large and charismatic species like elephants (van der Hoeven et al. 2004; Nash et al. 2016; Fig. 6.2). Permanent residents in the study area might be able to provide data on long-term trends and observations which could otherwise hardly be collected within the limited-time frame of average research projects. Interviews with local farmers were therefore used as a starting point for this thesis' field work to assess the extent and severity of conflicts as well as indications on habitat use of wildlife outside the natural forest.

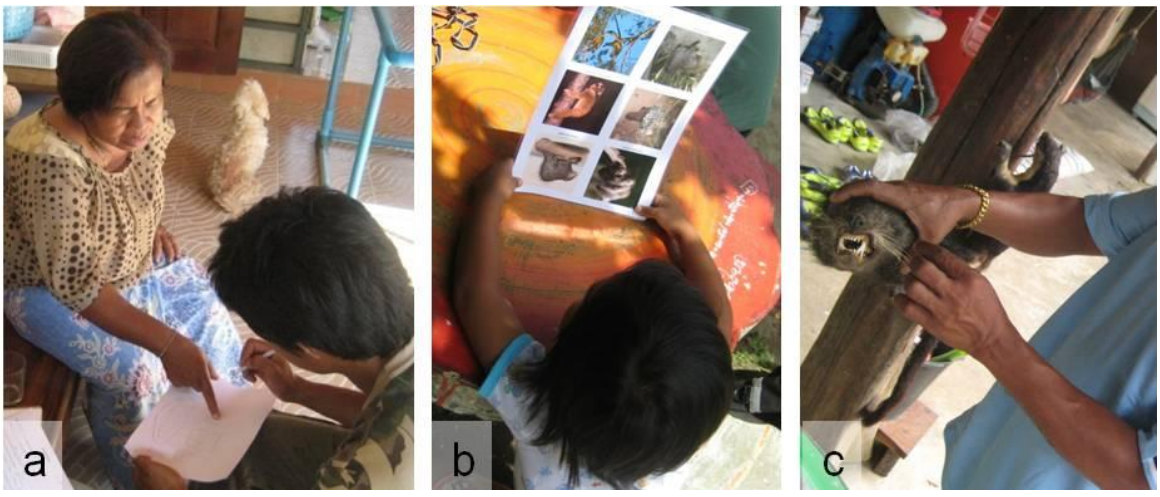

Figure 6.2. Interviews with local farmers around the TRY National Park: Woman explaining where elephants came to her farm as outlined on a sketched map (a), a farmer's child looking at animal pictures used to help respondents with the identification of wildlife species seen on their farms (b), and a farmer showing some of the wild animals he had hunted in the forest (c).

Questionnaire outcomes were complemented by transect and camera trap surveys. Camera traps have the advantage of being minimal invasive and of producing data on a continuous time scale (Kucera & Barrett 2011; O'Brien et al. 2011). They are efficient in detecting elusive, secretive or nocturnal species such as carnivores, albeit with varying detection probabilities for different size classes (Karanth et al. 2004; Kucera & Barrett 2011; Chutipong et al. 2014). Transect surveys further allowed the sampling of a broader environment (Buckland et al. 2008).

The results of our combined assessments highlighted the advantage of multi-method survey approaches (Table 6.1). The application of camera traps was more successful in capturing carnivore species than transect and interview surveys combined (Appendix E). In contrast, transect surveys were more effective in detecting primate species as the cameras were applied with a focus on the ground and cameras are generally less suitable for capturing arboreal species (O'Brien et al. 2011). Interviews produced lower species numbers but this was expected as the tropical and subtropical farmland represents less suitable wildlife habitat compared to natural forest (Martin & Sauerborn 2013).

Table 6.1. Assessment of species number through camera, transect and interview surveys.

Taxonomic group	**Species no.**	**Threatened status***	**Cameras** (Boundary/ Forest)	**Transects** (Forest/ Farmland)	**Interviews** (Farmland)
Ungulates	6	2	6	5	4
Primates	6	4	3	6	4
Carnivores	10	3	9	3	3
Rodents	8	0	7	7	6
Other	5	1	2	4	3
Total no.	**35**	**10**	**26**	**25**	**20**

*Species listed as threatened according to the IUCN red list classification

An important part of ecology is not only the study of the abundance, composition or distribution of wildlife populations but also their interactions with the environment (Buckland et al. 2001). Such interactions are for example the seed dispersal services of wild animals which contribute to forest diversity and carbon storage (Bello et al. 2015; Malhi et al. 2016). Megafauna such as elephants are particularly important for ecosystem

functioning (Campos-Arceiz & Blake 2011). The feeding and seed dispersal experiments in chapter 4 investigated the potential of the megaherbivores for the dispersal of *D. indica*, a tree species with characteristic megafaunal-syndrome fruits. As the results showed, this tree species seems to benefit from being eaten by elephants in terms of germination success and time. The large dispersal distances resulting from elephant fruit consumption might further gain increasing importance in times of climate change (Campos-Arceiz & Blake 2011; Corlett & Westcott 2013). Consequently, the reductions in natural habitat as well as the high levels of poaching and conflicts with people do not only threaten the survival of elephants and other large mammals but also the functions they fulfill and the services they provide in ecosystems.

In order to assess such ecosystem services, several analytical tools have been developed in recent times (Bagstad et al. 2013a). Among them are InVEst (Integrated Valuation of Ecosystem Services and Trade-offs; Sharp et al. 2016), ARIES (ARtificial Intelligence for Ecosystem Services; Villa et al. 2014) or MIMES (Multiscale Integrated Model of Ecosystem Services; Boumans et al. 2015). Nonetheless, suitable methods that account for the multiple facets of biodiversity assessments in agricultural areas remained a scarce aspect so far. The methodology introduced in chapter 5 allows the inclusion of various levels of species diversity into an integrated ESS evaluation model, in this case InVEst. This exemplary procedure is of significance as multi-species assemblages and their interactions with the environment are inextricably entwined with ecosystem processes (TEEB 2012). The approach is further suitable for scaling the results up to larger landscape and regional levels, which is often a major uncertainty in ecosystem service research (Loreau et al. 2001; Vihervaara et al. 2010).

6.2.2 Methodological limitations

A challenge in questionnaire surveys is the validation of the respondents' statements, a process that is often missing in ecological studies (White et al. 2005). Damage might be exaggerated and smaller or more elusive species might be less reported or wrongly identified (Mackenzie & Ahabyona 2012; Turvey et al. 2014; Nash et al. 2016). Where possible, i.e., in case of recent damage incidents, the respondents' damage claims were therefore verified on site. Camera and transect surveys additionally supported ground-truthing.

The difficult topography in our study site in southern Thailand hampered traditional line transect and distance sampling designs (Buckland et al. 2001). As walking straight lines over longer distances was impossible in the natural forest, we had to follow natural features such as wildlife trails or ridges (Buckland et al. 2008; Steinmetz et al. 2013). Therefore, we referred to species richness and presence assessments rather than to density and abundance parameters (MacKenzie et al. 2006). Transect surveys are a 'snapshot' method of a specific moment in time but repetitive sampling of transects as done in this study provides a larger time scale and the possibility to survey each site under different conditions (MacKenzie et al. 2006; Buckland et al. 2008). Camera traps in contrast provide continuous temporal data but capture only a limited range. Their successful application further depends on the cameras' locations and the species or species group of interest, e.g. small versus large-bodied species (Chutipong et al. 2014).

The focus of this study was on medium-sized to large mammals as this group represents a particularly important threatened part of vertebrate diversity (O'Brien et al. 2011). We nevertheless included the recorded data for smaller mammals, i.e., rodents, but the species number in this group is likely underrepresented as the identification down to species levels is difficult without live-trapping (Parmenter et al. 2003), an approach that was beyond the capacities of this thesis.

6.3 Ecological and socioeconomic aspects of wildlife presence and conservation in rubber-dominated landscapes

6.3.1 Trends in mammalian wildlife diversity and presence

Natural forests remain the major essential habitat for larger wildlife species, no matter how intensive the animals use the surrounding cultural landscape (Barlow et al. 2007; Gibson et al. 2011). Forests offer shelter during the daylight or serve as nursery grounds for the offspring (Gardner et al. 2009). Species richness is generally higher in natural tropical forests compared to cultural landscapes of cash crops such as rubber and oil palm as shown in the review in chapter 2. Distance from the forest edge is thereby often a strong predictor for declining biodiversity and conflict levels with people (Koh 2008; Gilroy et al. 2014; Yue et al. 2015; Crisol et al. 2016). This trend became very clear in the assessments of this thesis, where a decline in the total species number from the natural forest into the farmland

was shown while the species richness at the forest edge was still relatively high. Primates and smaller ungulates in particular showed a very limited dispersal range into the farmland. Wild pigs, for example, avoid areas without protective cover and their activity is typically restricted to ranges in close proximity to the forest edges (Schley et al. 2008; Thurfjell et al. 2009). Some arboreal primates on the other side rely on canopy connectivity or high understory vegetation for their movement and prefer taller trees while avoiding canopy gaps (Hopkins 2011; McLean et al. 2016). Many rodent species, in contrast, are well able to adapt to cultural landscapes (Morand et al. 2015), which was shown by their continuous albeit lower average presence in the farmland. Despite this supply in potential prey, carnivore signs were very scarce in the farmland. However, carnivores occur naturally in relatively lower abundances than prey species and are furthermore difficult to detect due to their elusive behavior (Karanth et al. 2004; Chutipong et al. 2014).
The average presence of elephants was much reduced in the farmland compared to the forest but the pachyderms nevertheless frequently entered the farmland. Signs of elephant presence were relatively abundant in the farm-forest transition zone (Fig. 6.3).

As it was beyond the time frame of this thesis to sample the entire conservation area for wildlife signs, uncertainty remains about the total elephant (or other wildlife species) population size in the Tai Rom Yen National Park (TRY) and the habitat use in the core zone of the forest. No scientific study on wildlife populations had previously been published from this area. Rigorous transect sampling or DNA probes would have been needed to obtain reliable density and population size estimates (Buckland et al. 2001; Hedges et al. 2013; Zhang et al. 2015). Therefore, it cannot unequivocally be established to what extent the pachyderms use the transition zone relative to the entire forest habitat. The difficult terrain inside the park partly characterized by limestone formations might nevertheless increase the attractiveness of the park edges, particularly in the dry season (Sukumar 1989; Thouless 1994; Sarker & Røskaft 2010; Bal et al. 2011). Surprisingly, hardly any elephant sightings were reported from the eastern side of the forest complex at the edges of the Khao Nun National Park, possibly another indication for the restricted forest use by the elephants. According to rangers and local residents, some few elephants might migrate from the South of TRY to Kra Toon Wildlife Sanctuary and Khao Luang National Park in the neighboring Nakhon Si Thammarat Province (Appendix F.1).

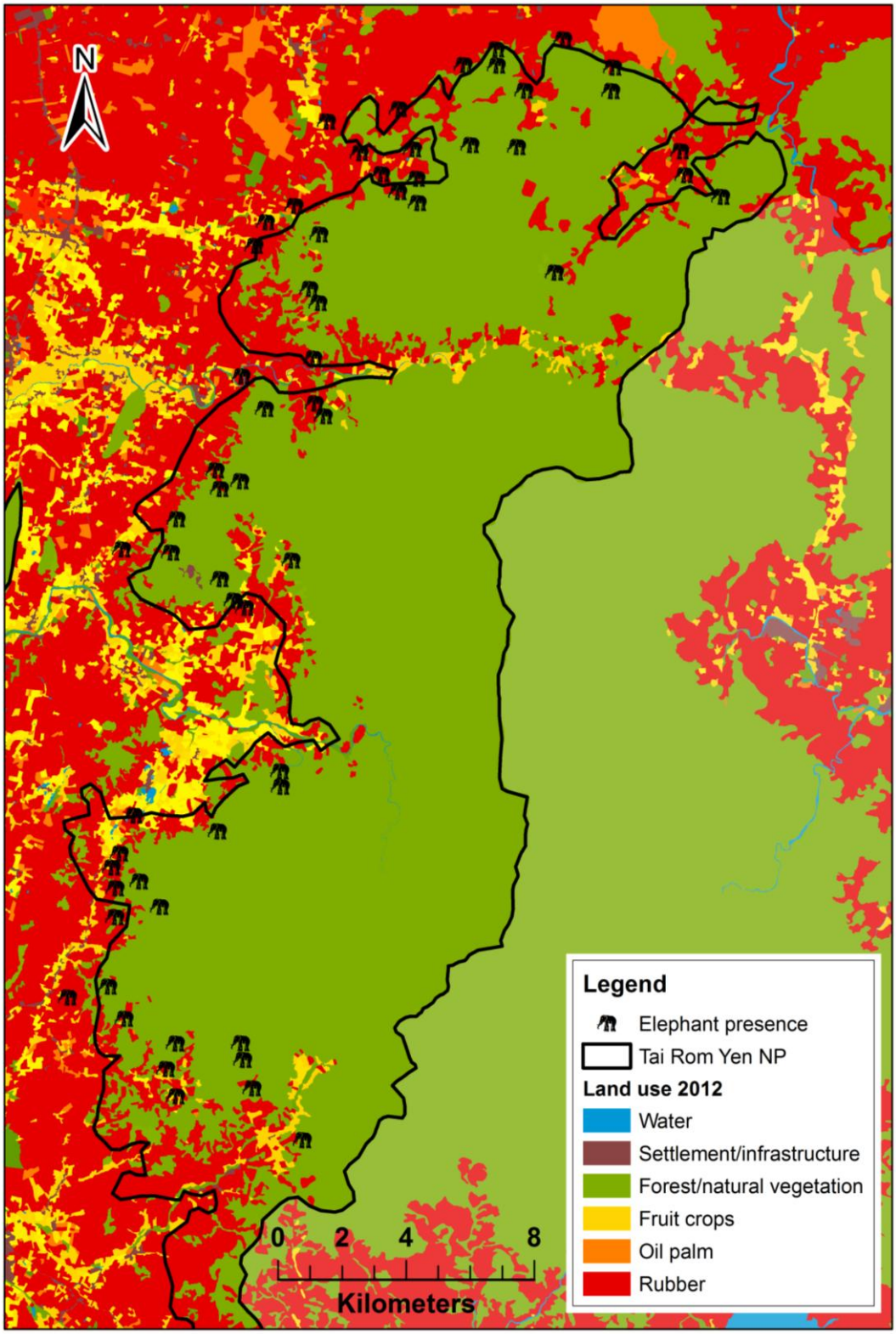

Figure 6.3. Elephant presence in and around the Tai Rom Yen National Park. The number of elephant symbols do not reflect the abundance but confirmed signs of presence. The land-use map was provided by the Land Development Department Thailand.

Like elsewhere in tropical and subtropical forests (e.g. Corlett 2007; Luskin et al. 2014; Ripple et al. 2016b), human encroachment and poaching levels were high in TRY. In fact, humans had a higher average presence in the forest than any wildlife species. Some people entering the forest collected non-timber forest products (NTFPs) other than wild animals (e.g. pods of wild *Parkia speciosa* used as vegetable) but half of the camera trap pictures displaying people could clearly be identified as poachers. Snares and gun shells were also frequently found during transect walks. This had likely strong implications on species richness and abundance in the area and could lead to further defaunation of large forest blocks in the long-term (Harrison 2011; Brodie et al. 2015).

6.3.2 Socioeconomic importance of wild mammals

Potential damage-causing species in rubber plantations include rats, porcupines, wild pigs, deer, primates and elephants (Jayarathnam 1992). However, rubber damage by mammals other than elephants was of no or negligible concern to most farmers around TRY. Relatively seen, overall crop loss by elephants affected only a minority of the farmers around TRY and plant diseases as caused for example by fungi were by far the most common cause for damage. Distance to the natural habitat is usually a strong predictor for wildlife damage likelihood as it was in this study and the economic importance of elephant damage is generally insignificant on a broader regional scale (Barnes et al. 1995; Naughton-Treves & Treves 2005; Sarker & Røskaft 2010; Wilson et al. 2015) . However, individual crop losses were sometimes very high, as is typical for damage by the megaherbivores (Naughton-Treves 1997). Some farmers further experienced damage to infrastructure such as cars, houses or fish ponds. Complaints might, however, in parts also be due to the elephants' status as a charismatic species protected by national and international law (Western & Waithaka 2005; Douglas & Veríssimo 2013). Damage by smaller species might much more often go unreported or deemed as irrelevant although the accumulated losses can be considerable as well (Lahm 1996; Arlet & Molleman 2010). Apart from the actual or perceived impact on crop harvest, elephants additionally pose a tangible threat to people's lives as they do not hesitate to attack if disturbed or even without direct provocation (Sukumar 1989) and 49% of interviewed farmers felt in danger from elephants. No human fatalities occurred during the time of this study but people had been killed by elephants in previous years around TRY.

An increased risk of encounter in the farmland of the study site might arise from the kind of rubber harvesting procedure. Rubber trees are tapped in the night or early morning hours when human sight is limited to the range of the flashlight. Elephants typically enter farmland between dusk and dawn (Sukumar 1989) and due to this overlap in human and animal activity, the chances of encounter might increase. Although damage to rubber trees in our study occurred in less than half of the cases when elephants visited farms, farmers sometimes had to restrain from harvesting rubber for a day or two, resulting in missed income opportunities. However, farmers were usually not too concerned by these short-time restrictions on rubber harvest as the trees are not tapped every day of the week anyway (Abraham & Joseph 2016). Also, in the case of strong rainfall harvesting is stopped due to operational challenges such as spillage or washout (Sanjeeva Rao & Vijayakumar 1992). Furthermore, rubber plants do not represent a particularly attractive food source to elephants and the trees are hardly affected by damage once they reach maturity at an age of around 5 to 8 years (Jayarathnam 1992; Chen et al. 2013). Young trees in contrast are highly susceptible, mainly as collateral damage but also as direct elephant feed, which caused economic losses of more than a million US$ in 2011 in Xishuangbanna in China (Chen 2012). However, this level of damage is unlikely to be reached in southern Thailand as this region has a long tradition in cultivating rubber and the proportion of young plantations is lower (Li & Fox 2012). In Xishuangbanna, where the rubber expansion was massive over the last years, close to 50% of rubber plantations were less than 8 years old (Chen 2012; Warren-Thomas et al. 2015). Around TRY only a third of interviewed farmers had patches of young plantations which often represented only a part of the farmer's entire rubber fields.

In contrast to other areas in Asia where wild elephant populations are a major visitor attraction (Tisdell & Zhu 1998; Buultjens et al. 2005; Zhang 2011; Aihara et al. 2016), tourism in and around TRY is very low and residents hardly benefit from any related revenues. The economic benefits some people obtain from the conservation area are often derived from officially prohibited resource extraction such as honey harvesting and collection of other NTFPs and the more severe illegal killing of wild animals for the own consumption or the wildlife trade. Hunting in many Southeast Asian countries nowadays tends to be primarily a commercial or cultural activity rather than a matter of subsistence (Corlett 2007; Luskin et al. 2014). Pangolins (e.g. *Manis javanica*) are in particularly high demand on the global market and a farmer in TRY confirmed to hunt this species and sell

the carcasses to China, one of the biggest market for pangolin meat and scales (Challender et al. 2015). Wild pigs and deer species further constitute major hunting targets but almost all species are at risk (Corlett 2007; Luskin et al. 2014). During field work for this thesis, evidence for a killed porcupine (*Hystrix brachyura*), binturong (*Arctictis binturong*) and leopard cat (*Prionailurus bengalensis*) was found and a local shop owner kept a juvenile gibbon (*Hylobates lar*) as a pet.

While the killing of elephants for their ivory is a major conservation concern in Asia, it does not reach current African poaching levels - some of the reasons being that the Asian elephant population is smaller and that female Asian elephants do not exhibit tusks while the male population in some areas has a high proportion of tuskless animals as well (Sukumar et al. 1998; Chelliah & Sukumar 2013). Also, in Thai culture, elephants have a very special role as royal animals and symbols of luck which are highly revered and killing elephants is therefore a cultural taboo (Ringis 1996; McNeely 2000). This might contribute to a relative high tolerance and positive attitude towards elephants shown by a large part of farmers around TRY.

Nevertheless, Thailand with its booming tourism industry is one of the major end-use markets for illegal African ivory, which is then further distributed globally (Milliken et al. 2012). The illicit ivory trade in China is even more than double as large as in Thailand (Milliken et al., 2012; Underwood et al. 2013). Elephant poaching levels in Africa were shown to be strongly correlated with the prices on the local black market and with an increase in ivory seizures with intended destination to China (Wittemyer et al. 2014). To reduce the demand, China's government announced a ban on the domestic ivory trade as well as all processing activities in December 2016 which will be implemented by the end of 2017 (China State Council 2016). The domestic market in Thailand is still open but the Thai government required traders and owners to register their ivory through an 'Elephant Ivory Act' passed in 2015 (Business Standard 2015).

6.3.3 Ecological functions and services of wildlife and of conservation areas

Farmers did not obtain any direct benefits from wild elephants but a vast majority (71%) appreciated the services provided by the natural forest. NTFPs such as fruits, vegetables and medicines play an important role in many people's basic necessities and can contribute

to the household income or to the reduction of livelihood risks (Angelsen et al. 2014; Sakai 2016). Animal protein derived from wildlife was certainly another benefit for some people, albeit rarely admitted during the interviews. Apart from providing NTFPs, forests play an important role for the water supply in terms of quantity, quality and temporal regulation (Brauman et al. 2007; Brogna et al. 2017) and the fresh water provisioning as well as the regional climate regulation were acknowledged by local farmers as ecosystem services of highest importance to them.

These services, however, are rarely derived ad hoc but are often the result of complex and interrelated processes (Brauman et al. 2007). Wildlife populations are part of this ecosystem complex and their decline or extinction might in the long-term lead to detrimental effects on the services valued by people, e.g. carbon storage or pollination (TEEB 2012; Malhi et al. 2016). Apart from their role in nutrient fluxes and lifecycle maintenance of many plants, wild mammals and particularly elephants support the habitat or food provisioning for smaller vertebrate and invertebrate species such as frogs and different families of beetles (Vinod & Sabu 2007; Campos-Arceiz 2009; Rattanawannee et al. 2013; Wolf et al. 2013). Natural forests with a high plant diversity as supported by elephants are of high importance for wild insects which again provide significant pollination services, also in agricultural systems bordering the forests (Ricketts et al. 2008; Campos-Arceiz & Blake 2011; Garibaldi et al. 2013; Cusser et al. 2016; Widhiono et al. 2016). Mammalian predators further support the control of potential pest species in agricultural landscapes: Leopard cats hunt rodents such as rats (e.g. *Rattus* spp. and *Maxomys* spp.) which do not only damage crops but also pose risks to human health (Rajaratnam et al. 2007; Morand et al. 2015). Despite being omnivorous with a preference for fruits, several civet species (e.g. *Paguma larvata*, *Paradoxurus hermaphrodites*) and hog badgers (*Arctonyx collaris*) prey on small vertebrate and invertebrate species as well (Joshi et al. 1995; Zhou et al. 2008; Zhou et al. 2015). The diet of insectivorous bats (e.g. *Hipposideros armiger*, *Taphozous longimanus*) includes pest moths (Lepidoptera) and mosquitoes (Culicidae), latter which are important human pathogen vectors (Reiskind & Wund 2009; Brown et al. 2015; Weterings et al. 2015).

The extent to which mammals can use the agricultural landscapes and simultaneously provide ecological services in the agroecosystems depends on the availability and distribution of resources which are governed by human land-use decisions (Daily 2001;

Barlow et al. 2007; Nakashima et al. 2013). As shown in the 'Go Green' land-use scenario in chapter 5, the habitat suitability of agricultural landscapes for three species groups increases with an improved protection of the ecologically most important land-use types, reforestation of areas less suitable for agriculture (i.e., steep slopes) and enrichment of rubber plantations (Cotter et al. 2014).

6.4 Future steps for more sustainable and conflict-reduced rubber cultivation

Rubber cultivation simultaneously holds challenges and chances for the development of more sustainable land-use practices. The replacement of natural forests into rubber plantations leads to declines in biodiversity and carbon losses but the conversion of arable land into rubber plantations might have positive environmental effects in terms of carbon sequestration (Blagodatsky et al. 2016). However, the general cultivation practice as a monoculture combined with clean-weeding through pesticide application leads to species-poor environments and can have negative impacts on the ground and surface water quality (Arias-Estévez et al. 2008; Langenberger et al. 2016; Liu et al. 2016). A major challenge in the quest for more sustainability is therefore to promote reduced pesticide use and to support an increased heterogeneity within the plantations.

Permanent intercropping is theoretically a promising approach to enrich the monoculture plantations but is rarely adopted by farmers due to the work load or missing economic incentives (Langenberger et al. 2016). Furthermore, any proposed intercrop species in proximity to natural elephant habitat should be evaluated on its attractiveness for the megaherbivores in order to avoid conflicts. High-value timber species could be an interesting crop candidate (Somboonsuke et al. 2011) but would require extra protection against elephants in the establishment years when the trees are young and susceptible. As intercropping is seemingly not lucrative enough to be adopted by farmers on a large scale (Langenberger et al. 2016), the approximately 7 m broad gap between rubber rows could be left for natural undergrowth, with only a clear-cut pathway along the rubber tree lines to allow farmers easy access for tapping. Natural ground cover can thus simultaneously contribute to erosion control, soil health and to a certain extent to biodiversity conservation (Aratrakorn et al. 2006; Abraham & Joseph 2016; Liu et al. 2016).

Several farmers around TRY maintained the natural flora in between the rubber rows to some extent (Fig. 6.4). Some of these farmers reported that elephants came to their farm and fed on the undergrowth vegetation such as wild banana (*Musa acuminata*), a plant that is used as feed source by a wide range of other wildlife species as well (Marod et al. 2010). Green biomass serves as resource that drives higher trophic cascades (Hunter & Price 1992). Non-crop vegetation can further serve as shelter for wildlife and a more diverse environment likely benefits biological pest control (Bianchi et al. 2006; Harvey et al. 2006).

A challenge in the promotion of increased natural vegetation is the farmers' traditionally critical view on undergrowth, which is often deemed as competition to crops and feared as source for poisonous animals such as some caterpillars or snakes (Abraham & Joseph 2016; Langenberger et al. 2016; Liu et al. 2016). Making their farms attractive to wild mammals might also often not be in the particular interest of farmers as some species such as wild pigs or elephants can pose real dangers.

Figure 6.4. Four examples of different farm management: On the left side farms where all undergrowth was cut or sprayed with pesticides while the plantation owners on the right side allow some natural vegetation to persist.

It is therefore not sure if a majority of farmers would consider the benefits of keeping higher levels of natural vegetation as more convincing than real or perceived negative implications. A potential motivation could be that many farmers are well aware of various services provided by ecosystems such as fresh water, particularly if they are living in proximity to natural areas (Zhang et al. 2016). With deteriorating services of high importance such as safe water supply but also erosion and flood control, more and more people are likely to understand the environmental problems of certain land-use developments (Ahlheim et al. 2015). Awareness might nevertheless not be enough to stimulate altered individual long-term management approaches which are strongly market-driven (Zhou 2000; Zhang et al. 2016). This might change if the market starts to demand more sustainably derived products. The International Rubber Study Group (IRSG; http://www.rubberstudy.com/), an inter-governmental organization comprising rubber producing as well as consuming stakeholders recently launched a Sustainable Natural Rubber initiative (SNRi). The initiative's aim is to "establish the sustainability of the natural rubber value chain" (http://www.snr-i.org/). Among the proposed drivers for the voluntary implementation of the guidelines are the prospects of increased yield, reduced production costs and the avoidance of wastage leading in return to higher income for the producers (IRSG 2014). As this initiative is voluntary, self-certified and the vaguely defined targets rather focus on business than environmental sustainability, its effectiveness remains to be seen (Langenberger et al. 2016). More promising would be a scheme that adheres to clearly defined and strict regulations controlled by an independent party and which would include financial incentives as is the case in organic agriculture (e.g. Sgroi et al. 2015; Hardman et al. 2016; Qiao et al. 2016). Such an initiative would include rewards for wildlife-friendly management approaches (Hardman et al. 2016) that could also help to compensate wildlife damage. However, such "organic rubber" production has yet to be developed and implemented.

In the meantime, conflicts with wildlife and particularly with elephants remain a major threat to conservation efforts and therefore any management plan needs to include potential mitigation approaches (Fernando et al. 2008). Translocation of elephants as frequently suggested by farmers is no option, as elephant numbers in TRY are likely very low already and as alternative suitable habitat is scarce. Additionally, the risks for the involved animals and people handling them are immense and translocations have negative impacts on the social network of the elephant population (Fernando et al. 2012; de Silva et al. 2016).

Insurance schemes can help to improve the acceptance of people sharing the same environment with wildlife (Chen et al. 2013) but without a respective insurance company willing to invest in the area or insufficient funds of government authorities, the realization is difficult.

Feasible options should be of relatively low costs and easy application (Fernando et al. 2008). An approach to minimize the risk of accidental encounters could be non-electric alarm fences equipped with noise-making objects that are set up at the edge of fields bordering the forest (Osborn & Parker 2003; Fernando et al. 2008). Such alarm fences with sirens and light triggers are successfully combined with community guarding in high risk areas around the Kaeng Krachan National Park in Thailand (personal observation; Fig. 6.5). As soon as the elephants try to break through the fence to feed on crops such as pineapples, the noise and light is triggered and farmers on guarding duty start to throw firecrackers to drive the animals back to the forest. In the case of mature rubber plantations where not damage but accidental encounters are the risk, alarm fences alone might be sufficient to warn the farmers of the elephants' presence.

An additional alarm system could be set up using social media. A farming community in Malaysia successfully established a 'WhatsApp' group for elephant alerts through which members can warn each other when elephants have been observed in the farmland (The Star Online 2016). This could be applied in Thailand as well where social media use is widespread and was already intensively used during the flood crisis in 2011 to exchange the most recent information on the flood situation (Leong et al. 2015). A limitation might however arise from gaps in the mobile phone signal coverage in some of the more remote areas around the park.

Guarding crops is time-intensive and therefore only likely to be conducted by farmers in areas with frequent and high economic losses due to elephants (Sarker & Røskaft 2010). Around TRY, 20 farmers resorted to guarding their crops, albeit mostly not to protect rubber but fruit orchards. Half of these farmers only guarded their crops during the harvest season when the fruits were ripe. For farmers in elephant high risk areas, i.e. in proximity to the forest edge, it might be advisable to focus on rubber cultivation and resign from fruit crop cultivation altogether if frequent damage occurs (Osborn & Parker 2003).

In the case of young rubber plantations, alarm fences would not be of much use as - unlike the limited high risk phase in fruit crops - the young plants are susceptible to elephant damage over an extended period of approximately 5 to 8 years (Paardekooper 1989; Chen et al. 2013). Here, electric fences might be a better option as they can provide effective protection against elephants (Sukumar 1989; Kioko et al. 2008). The maintenance issues of electric fences (Graham et al. 2009; Hoare 2012) might not be a problem if the fence is in the private responsibility of individual smallholder farmers who fence a relatively small area around young tree plantations. A higher obstacle are probably the associated costs, which will make electric fences only an economically viable option in very high risk areas where damage frequently occurs (Kioko et al. 2008; Graham et al. 2009). A cheaper option could be simple string fences on which a mix of chili, tobacco and old engine oil is applied to (Sitati & Walpole 2006; Chelliah et al. 2010). These chili grease fences would not stop elephants from entering but might evoke the choice of different routes through the farmland as young rubber plants do not constitute a targeted feed crop per se. Electric fences are also problematic if not properly installed using a fence controller due to the risk of electrocution. As a consequence of the death of two elephants that were killed through electric fences in TRY in 2013, the local NP authorities discourage farmers from setting up electric fences.

Figure 6.5. Guarding hut in proximity to an alarm fence erected at the farm-forest boundary of the Kaeng Krachan NP in Thailand. When elephants approach the fence, farmers throw firecracker to drive the animals back to the forest.

Using only wildlife deterrent measures to reduce conflicts is just like throwing "aspirin at elephants" (Barnes 2002) as they do not tackle the underlying causes of the problem, i.e., the extensive modification of landscapes accompanied by diminishing natural wildlife habitats. The stringent protection of the remaining natural habitat also outside of protected areas is therefore of crucial importance in the endeavor to reduce conflicts. Improving the connectivity of forest blocks is a key part of this strategy (Goswami & Vasudev 2017). In order to enhance the resource availability for wild animals, forests could be enriched with attractive local tree species (e.g. *D. indica*), artificial salt licks and water sources (Waitkuwait 1992; Parren et al. 2002; Zhang & Wang 2003; Fig. 6.6).

No single measure will be sufficient to reduce the conflicts between human land-use and wildlife populations emerging from conservation areas. The most promising approach lies in combining multiple deterrent tools with long-term land-use and conservation management. In any case, there will be no cure to eliminate the conflicts altogether as long as people and wildlife have to share limited resources. The best hope for a peaceful coexistence lies in reducing the conflicts to levels tolerable for both people and wildlife. There are still major challenges to address but at least in the case of the rubber-dominated landscape in southern Thailand, achieving such levels seems possible.

Figure 6.6. Wild elephant bull at an artificial salt lick in Tai Rom Yen NP.

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Appendices

Appendix A

Table A1. Species recorded in rubber and oil palm plantations. CR = Critically Endangered, E = Endangered, NT = Near Threatened, V = Vulnerable, LC = Least Concern, DD = Data Deficient.; p.c. = personal communication. (+) for rubber indicates that species were reported from Indonesia, where jungle rubber plantations are common, (+) for oil palm indicates that species were only recorded from locations within a limited distance to natural forest (up to 5km and 13km, respectively).

Family	Species	Common name	Red List Status	rubber	oil palm	References
Ungulates						
Elephantidae	*Elephas maximus*	Asian elephant	E	+	+	(32,33,35,46,54,57,60, 149)
Suidae	*Sus scrofa*	Wild pig	LC	+	+	(29,42,46,53,59,63,66)
Suidae	*Sus barbatus*	Bearded Pig	V		+	(42,62,72)
Tragulidae	*Tragulus kanchil*	Lesser Mousedeer	LC		+	(63,72)
Tragulidae	*Tragulus napu*	Greater Mousedeer	LC		+	(63,72)
Cervidae	*Muntiacus muntjak*	Red Muntjac	LC		+	(42,72)
Cervidae	*Muntiacus atherodes*	Bornean Yellow Muntjac	LC		+	(72)
Cervidae	*Rusa unicolor*	Sambar	V		+	(42)
Tapiridae	*Tapirus indicus*	Malayan Tapir	E		(+)	(63)
Carnivores						
Felidae	*Panthera tigris*	Tiger	E	(+)	(+)	(58,63,85, 148, 149)
Felidae	*Panthera pardus*	Leopard	NT		+	(63)
Felidae	*Prionailurus bengalensis*	Leopard Cat	LC		+	(42,62,63,69,72)
Felidae	*Prionailurus planiceps*	Flat-headed Cat	E		+	(63)

Felidae	*Neofelis nebulosa*	Clouded Leopard	V		+	(63)
Viverridae	*Arctictis binturong*	Binturong	V		+	(63)
Viverridae	*Arctogalidia trivirgata*	Small-toothed Palm Civet	LC		+	(63)
Viverridae	*Paguma larvata*	Masked Palm Civet	LC		+	(63)
Viverridae	*Viverricula indica*	Small Indian Civet	LC	+	+	(29,63)
Viverridae	*Viverra tangalunga*	Malay Civet	LC			(63,69,72)
Viverridae	*Viverra zibetha*	Large Indian Civet	NT		+	(63)
Viverridae	*Hemigalus derbyanus*	Banded Civet	V		+	(63,72)
Viverridae	*Paradoxurus hermaphroditus*	Common Palm Civet	LC	+	+	(29,42,44,53,61-63, 69,72)
Prionodontidae	*Prionodon linsang*	Banded Linsang	LC		+	(63)
Herpestidae	*Herpestes brachyurus*	Short-tailed Mongoose	LC	+	+	(29,72)
Herpestidae	*Herpestes smithii*	Ruddy Mongoose	LC		+	(29,53)
Herpestidae	*Herpestes semitorquatus*	Collared Mongoose	DD		+	(72)
Mephitidae	*Mydaus javanensis*	Malay Badger	LC		+	(72)
Mustelidae	*Lutra lutra*	Eurasian Otter	NT	+	+	(53)
Mustelidae	*Lutrogale perspicillata*	Smooth-coated Otter	V		+	(42)
Canidae	*Canis aureus*	Golden Jackal	LC	+	+	(29,53)
Ursidae	*Helarctos malayanus*	Sun bear	V		+	(42,63)
Primates						
Hominidae	*Pongo abelii*	Sumatran Orangutan	CR	+	+	(46,48,55,67)
Hominidae	*Pongo pygmaeus*	Bornean Orangutan	E		+	(64)
Hylobatidae	*Hylobates lar*	White-handed Gibbon	E		+	(63)
Hylobatidae	*Symphalangus syndactylus*	Siamang	E		+	(63)
Cercopithecidae	*Macaca fascicularis*	Long-tailed Macaque	LC		+	(42,43,46,59,63,72)
Cercopithecidae	*Macaca mulatta*	Rhesus Macaque	LC	+		(38)
Cercopithecidae	*Macaca nemestrina*	Southern Pig-tailed Macaque	V		+	(42,63,72)

Cercopithecidae	*Macaca sinica*	Toque monkey	E		+	(53)
Cercopithecidae	*Presbytis melalophos*	Sumatran Surili	E	+	+	(28,42)
Cercopithecidae	*Presbytis thomasi*	Thomas' Leaf Monkey	V	+		(46)
Cercopithecidae	*Trachypithecus geei*	Gee's Golden Langur	E	+		(38)
Cercopithecidae	*Trachypithecus cristatus*	Silvery Lutung	NT		+	(28,59,63)
Cercopithecidae	*Trachypithecus vetulus*	Purple-faced leaf monkey	E	+	+	(29,41,49,56)
Cercopithecidae	*Trachypithecus obscurus*	Dusky Leaf-monkey	NT		+	(63)
Lorisidae	*Nycticebus coucang*	Greater Slow Loris	V		+	(42,63)
Bats						
Emballonuridae	*Taphozous longimanus*	Long-winged Tomb Bat	LC	+		(50)
Emballonuridae	*Emballonura monticola*	Lesser Sheath-tailed Bat	LC	+		(50)
Hipposideridae	*Hipposideros bicolor*	Bicolored Leaf-nosed Bat	LC	+		(50)
Hipposideridae	*Hipposideros cervinus*	Fawn Leaf-nosed Bat			+	(45)
Hipposideridae	*Hipposideros larvatus*	Horsfield's Leaf-nosed Bat	LC	+		(50)
Megadermatidae	*Megaderma lyra*	Greater False Vampire	LC		+	(52)
Megadermatidae	*Megaderma spasma*	Lesser False Vampire	LC	+	+	(29,50,53)
Miniopteridae	*Miniopterus magnater*	Large Bent-winged Bat	LC	+		(50)
Pteropodidae	*Cynopterus brachyotis*	Lesser Short-nosed Fruit Bat	LC	+	+	(28,45,52,68)
Pteropodidae	*Cynopterus horsfieldii*	Horsfield's Fruit Bat	LC	+	+	(28,52)
Pteropodidae	*Cynopterus sphinx*	Greater Shortnosed Fruit Bat	LC		+	(52)
Pteropodidae	*Eonycteris spelaea*	Dawn Bat	LC		+	(45,52,68)
Pteropodidae	*Pteropus giganteus*	Indian flying-fox	LC	+	+	(29)
Pteropodidae	*Rousettus amplexicaudatus*	Geoffroy's Rousette	LC		+	(52)
Pteropodidae	*Rousettus leschenaultii*	Leschenault's Rousette	LC	+	+	(29)
Rhinolophidae	*Rhinolophus affinis*	Inter-mediate Horseshoe Bat	LC	+		(50)
Rhinolophidae	*Rhinolophus lepidus*	Blyth's Horseshoe Bat	LC	+		(50)

Rhinolophidae	*Rhinolophus luctus*	Woolly Horsehoe Bat	LC	+		(50)
Rhinolophidae	*Rhinolophus robinsoni*	Peninsular Horseshoe Bat	NT	+		(50)
Rhinolophidae	*Rhinolophus rouxi*	Rufous horse-shoe bat	LC	+	+	(29)
Rhinolophidae	*Rhinolophus stheno*	Lesser Brown Horseshoe Bat	LC	+		(50)
Rhinolophidae	*Rhinolophus trifoliatus*	Trefoil Horseshoe Bat	LC	+		(50)
Vespertilionidae	*Pipistrellus tenuis*	Least Pipistrelle	CR	+	+	(29)
Vespertilionidae	*Phoniscus jagorii*	Peters's Trumpet-eared Bat	LC	+		(50)
Vespertilionidae	*Scotophilus kuhlii*	Lesser Asiatic Yellow Bat	LC		+	(52)
Rodents						
Hystricidae	*Atherurus macrourus*	Asiatic Brush-tailed Porcupine	LC		+	(63)
Hystricidae	*Hystrix indica*	Indian Crested Porcupine	LC	+	+	(29,53)
Hystricidae	*Hystrix brachyura*	Malayan Porcupine	LC		+	(42,46,59,63,72)
Hystricidae	*Hystrix crassispinis*	Thick-spined Porcupine	LC		+	(72)
Hystricidae	*Trichys fasciculata*	Long-tailed Porcupine	LC			(72)
Sciuridae spp					+	(46)
Sciuridae	*Funambulus palmarum*	Common Palm-squirrel	LC	+	+	(29,53)
Sciuridae	*Callosciurus notatus*	Plantain Squirrel	LC	+	+	(40,71)
Sciuridae	*Callosciurus orestes*	Borneo Black-banded Squirrel	LC		+	[1]PT. REA Kaltim Plantations
Sciuridae	*Callosciurus prevostii*	Prevost's Squirrel	LC		+	(42)
Sciuridae	*Ratufa macroura*	Giant squirrel	NT	+		(53)
Sciuridae	*Sundasciurus lowii*	Low's Squirrel	LC		+	[1]PT. REA Kaltim Plantations
Sciuridae	*Sundasciurus tenuis*	Slender Squirrel	LC		+	[1]PT. REA Kaltim Plantations
Muridae	*Bandicota indica*	Greater Bandicoot Rat	LC	+	+	(29,53)
Muridae	*Mus musculus*	House mouse	LC	+	+	(29,40)
Muridae	*Mus caroli*	Ricefield Mouse	LC	+		(p.c., Serge Morand, 2015)
Muridae	*Mus cervicolor*	Fawn-colored Mouse	LC	+		(p.c., Serge Morand, 2015)

Muridae	*Mus cookii*	Cook's Mouse	LC	+		(p.c., Serge Morand, 2015)
Muridae	*Niviventer cremoriventer*	Dark-tailed Tree Rat	V	+	+	(40,93)
Muridae	*Niviventer fulvescens*	Chestnut White-bellied Rat	LC	+		(p.c., Serge Morand, 2015)
Muridae	*Maxomys rajah*	Rajah Spiny Rat	V	+		(40)
Muridae	*Maxomys surifer*	Red Spiny Rat	LC		+	(44,93)
Muridae	*Maxomys whiteheadi*	Whitehead's Spiny Rat	V	+	+	(40,44,93)
Muridae	*Sundamys muelleri*	Müller's Giant Sunda Rat	LC	+	+	(40,44)
Muridae	*Rattus argentiventer*	Rice-field Rat	LC		+	(51)
Muridae	*Rattus exulans*	Polynesian Rat	LC	+		(40)
Muridae	*Rattus losea*	Losea Rat	LC	+		(p.c., Serge Morand, 2015)
Muridae	*Rattus rattus*	House rat	LC	+	+	(29,40,51)
Muridae	*Rattus tiomanicus*	Malaysian Field Rat	LC	+	+	(40)
other small mammals						
Erinaceidae	*Echinosorex gymnura*	Moonrat	LC	+	+	(40,42)
Tupaiidae	*Tupaia glis*	Common Treeshrew	LC		+	(42)
Tupaiidae	*Tupaia minor*	Lesser Treeshrew	LC	+		(40)
Tupaiidae	*Tupaia splendidula*	Ruddy Treeshrew	LC	+		(40)
Tupaiidae	*Tupaia tana*	Large Treeshrew	LC	+		(40)
Manidae	*Manis crassicaudata*	Indian Pangolin	E	+		(53)
Manidae	*Manis javanica*	Sunda Pangolin	CR		+	(42,59,59,63,63,72)
Leporidae	*Lepus nigricollis*	Indian hare	LC	+	+	(29,53)
Cynocephalidae	*Galeopterus variegatus*	Sunda Flying Lemur	LC		+	(63)
Total species				57	80	

[1]PT. REA Kaltim Plantations Conservation Department 2008-2012, East Kalimantan, Indonesia

Appendix B

Table B1. List of wildlife species recorded during transect surveys, through camera traps and reported by farmers (1 = present, 0 = absent). Species found in the farmland are displayed with and without data from interviews. IUCN status CR = Critically endangered, EN = endangered, VU = vulnerable, NT = Near threatened, LC = Least concern.

Group	Order	Family	Species common	Species scientific	IUCN status	Forest	boundary	farmland	farmland + interviews
Ungulates	Proboscidea	Elephantidae	Asian elephant	*Elephas maximus*	EN	1	1	1	1
	Artiodactyla	Suidae	Eurasian wild pig	*Sus scrofa*	LC	1	1	1	1
		Tragulidae	Lesser mousedeer	*Tragulus kanchil*	LC	1	1	0	1
			Greater mousedeer	*Tragulus napu*	LC	1	1	0	1
		Cervidae	Red muntjak	*Muntiacus muntjak*	LC	1	1	0	0
		Bovidae	Southern serow or Chinese serow	*Capricornis sumatraensis/ C. milneedwardsii*	VU/ NT	1	1	0	0
Primates	Primates	Cercopithecidae	Langur	*Trachypithecus obscurus/Presbytis femoralis*	NT	1	0	0	0
			Long-tailed macaque	*Macaca fascicularis*	LC	1	0	0	1
			Southern pig-tailed macaque	*Macaca nemestrina*	VU	1	1	0	1
			Stump-tailed macaque	*Macaca arctoides*	VU	1	1	0	1

		Hylobatidae	White-handed gibbon	*Hylobates lar*	EN	1	0	0	0
		Lorisidae	Sunda slow loris	*Nycticebus coucang*	VU	1	0	1	1
Carnivores	Carnivora	Mustelidae	Hog badger	*Arctonyx collaris*	NT	1	1	0	0
			Yellow-throated marten	*Martes flavigula*	LC	1	0	0	0
		Viverridae	Large Indian civet	*Viverra zibetha*	NT	1	1	1	1
			Masked palm civet	*Paguma larvata*	LC	1	1	0	1
			Banded civet	*Hemigalus derbyanus*	VU	1	0	0	0
			Banded linsang	*Prionodon linsang*	LC	1	0	0	0
			Binturong	*Arctictis binturong*	VU	1	0	0	0
		Herpestidae	Crab-eating mongoose	*Herpestes urva*	LC	0	1	0	0
		Felidae	Leopard cat	*Prionailurus bengalensis*	LC	1	1	0	1
			Marbled cat	*Pardofelis marmorata*	VU	1	1	0	0
Rodents	Rodentia	Sciuridae	Black giant squirrel	*Ratufa bicolor*	NT	1	0	0	0
			Himalayan striped squirrel	*Tamiops macclellandii*	LC	1	1	1	1
			Slender squirrel	*Sundasciurus tenuis*	LC	1	1	1	1
		Muridae	Long-tailed giant rat	*Leopoldamys sabanus*	LC	1	1	0	0

			Several unidentified rat and mice species	*Rattus spec., Mus spec.*		1		1	1
		Spalacidae	Indomalayan bamboo rat	*Rhizomys sumatrensis*	LC	1	1	0	1
		Hystricidae	Malayan porcupine	*Hystrix brachyura*	LC	1	1	0	1
			Asiatic Brush-tailed Porcupine	*Atherurus macrourus*	LC	1	1	0	1
Other mammals	Erinaceomorpha	Erinaceidae	Moonrat	*Echinosorex gymnurus*	LC	1	1	0	0
	Scandentia	Tupaiidae	Common treeshrew	*Tupaia glis*	LC	1	1	1	1
	Chiroptera	Pteropodidae	Bats	*Cynopterus sp.*	LC	1	0	1	1
	Pholidota	Manidae	Sunda pangolin	*Manis javanica*	CR	1	1	0	1
	Dermoptera	Cynocephalidae	Sunda colugo	*Galeopterus variegatus*	LC	1	1	0	0

Appendix C

Table C.1. Summary of the number of censored and uncensored values.

Stratum	Categories	Total	Germinated	Censored	Percent Censored
1	**Control**	300	184	116	38.67
2	**GPT ≤ 30 h**	300	207	93	31.00
3	**30h < GPT ≤ 48 h**	410	274	136	33.17
4	**GPT > 48 h**	190	152	38	20.00
Total		1200	817	383	31.92

Table C.2. Wilcoxon trend test for the null hypothesis that the expected (mean) time to germination for the four different categories of seeds are the same versus an ordered alternative.

χ^2	*SE*	*z-Score*	*P* > |*z*|	*P* < *z*	*P* > *z*
186860	24053	7.7687	7.93×10^{-15}	1.00	3.96×10^{-15}

Table C.3. Tests for homogeneity in the germination time functions among categories.

Adjustment for Multiple Comparisons for the Wilcoxon Test				
Strata Comparison		χ^2	**P-Values**	
Categories	**Categories**		**Raw**	**Simulated**
Control	Short	22.2	<0.0001	<0.0001
Control	Medium	13.6	0.0002	0.0008
Control	Long	76.7	<0.0001	<0.0001
Short	Medium	0.4	0.5027	0.9091
Short	Long	11.5	0.0007	0.0029
Medium	Long	15.2	<0.0001	0.0002

Table C.4. Pairwise contrasts of the strength of the association between the time to germination and GPT categories and dung treatment. The contrasts are adjusted for multiplicity using simulation adjustment.

No	Comparisons	Estimate	SE	z - Value	Prob > |z|	AdjP	Adj Lower	Adj Upper
1	Control vs. Short	0.3366	0.0642	5.2423	<0.0001	0	0.2112	0.4620
2	Control vs. Medium	0.3589	0.0603	5.9565	<0.0001	0	0.2405	0.4773
3	Medium vs. Long	0.0223	0.0595	0.3748	0.7078	0.7080	-0.0944	0.1389
4	Control vs. Long	0.5879	0.0720	8.1608	<0.0001	0	0.4455	0.7302
5	Short vs. Long	0.2513	0.0713	3.5251	0.0004	0.0006	0.1131	0.3894
6	Medium vs. Long	0.2290	0.0682	3.3573	0.0008	0.0008	0.0939	0.3641
7	No Dung vs Dung	-0.3567	0.0451	-7.9039	<0.0001	0	-0.4455	-0.2679

Table C.5. Univariate Wilcoxon and log-rank tests for assessing if the effects of dung treatment and planting date pooled over the four strata (categories) are associated with germination time.

	Variable	*Test Statistic*	*SE*	χ^2	*P* > χ^2
Wilcoxon	Dung	74.5	9.7	58.6	<0.0001
	Date	2608.1	194.2	180.3	<0.0001
Log-rank	Dung	88.0	14.1	38.8	<0.0001
	Date	2440.6	285.2	73.2	<0.0001

Table C.6. Wilcoxon and log-rank tests for assessing whether dung treatment and planting date are associated with germination time based on forward stepwise sequence of χ^2. DF= Degrees of freedom.

	Variable	DF	χ^2	$P > \chi^2$	$\Delta\chi^2$	$P > \Delta\chi^2$
Wilcoxon	Date	1	180.3	<0.0001	180.3	<0.0001
	Dung	2	254.3	<0.0001	74.0	<0.0001
Log-Rank	Date	1	73.2	<0.0001	73.2	<0.0001
	Dung	2	126.7	<0.0001	53.5	<0.0001

Table C.7. Statistics for testing the association between germination time and categories, dung treatment and date of planting seeds. DF= Degree of freedom.

Variable	DF	Wald χ^2	P > χ^2
Categories	3	72.64035	<0.0001
Dung	1	62.47095	<0.0001
Date	1	140.8654	<0.0001

Appendix D

Habitat Quality Modeling, InVEST 3.2.0

For detailed information on the model please refer to InVEST User Guide, Chapter Habitat Quality (www.naturalcapital.org). For the model runs, we used the following input data.

Current and future land cover

Our main research area is the Naban River Watershed National Nature Reserve (NRWNNR) located in the Dai Autonomous Prefecture of Xishuangbanna (Figure D.1).

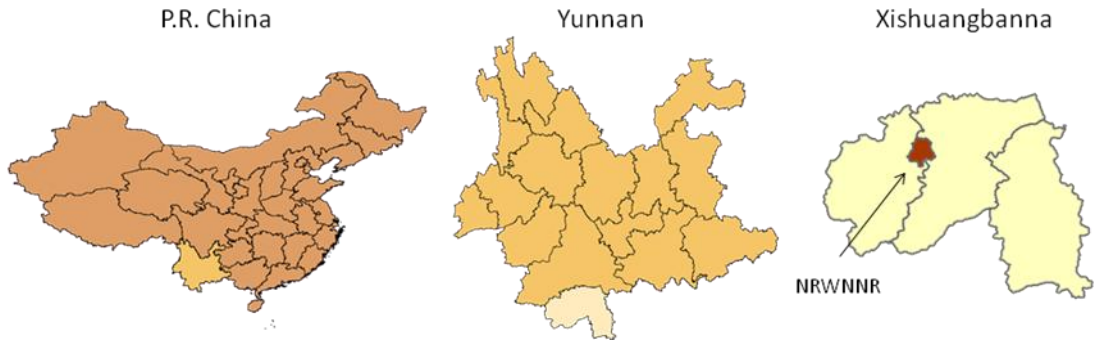

Figure D.1. Location of Research area. From left to right: PR China with highlighted Province of Yunnan, Province of Yunnan with highlighted Dai Autonomous Prefecture of Xishuangbanna, Dai Autonomous Prefecture of Xishuangbanna with highlighted research area (Global Administrative Area, http://www.gadm.org/).

The habitat quality model requires a current land cover which was available from the Project Living Landscapes China (Lilac) from 2007 (Wehner et al. 2014). As future land covers we used the two scenarios Business as usual (Gibreel et al. 2014) and Go Green (Cotter et al. 2014), both calculated for the year 2025, which were also available from Lilac (Figure D3 and Figure D4). For all three maps, land use classes were simplified by reclassification to fit the current research question (Figure D2-4).

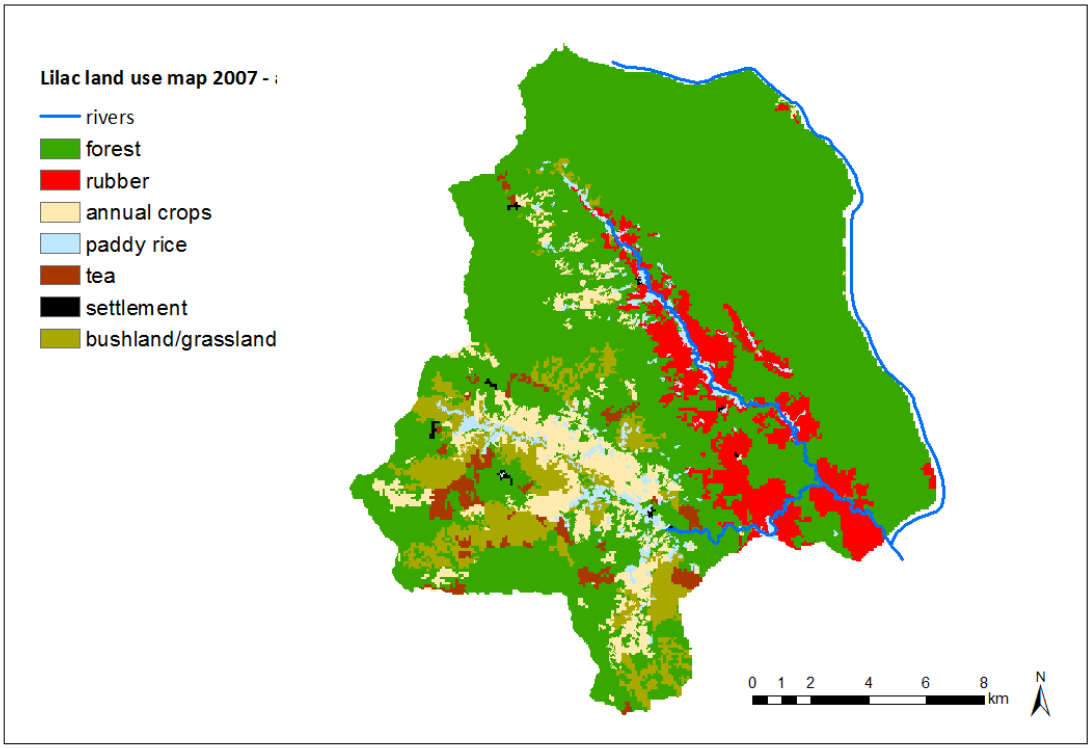

Figure D.2. Current land use map

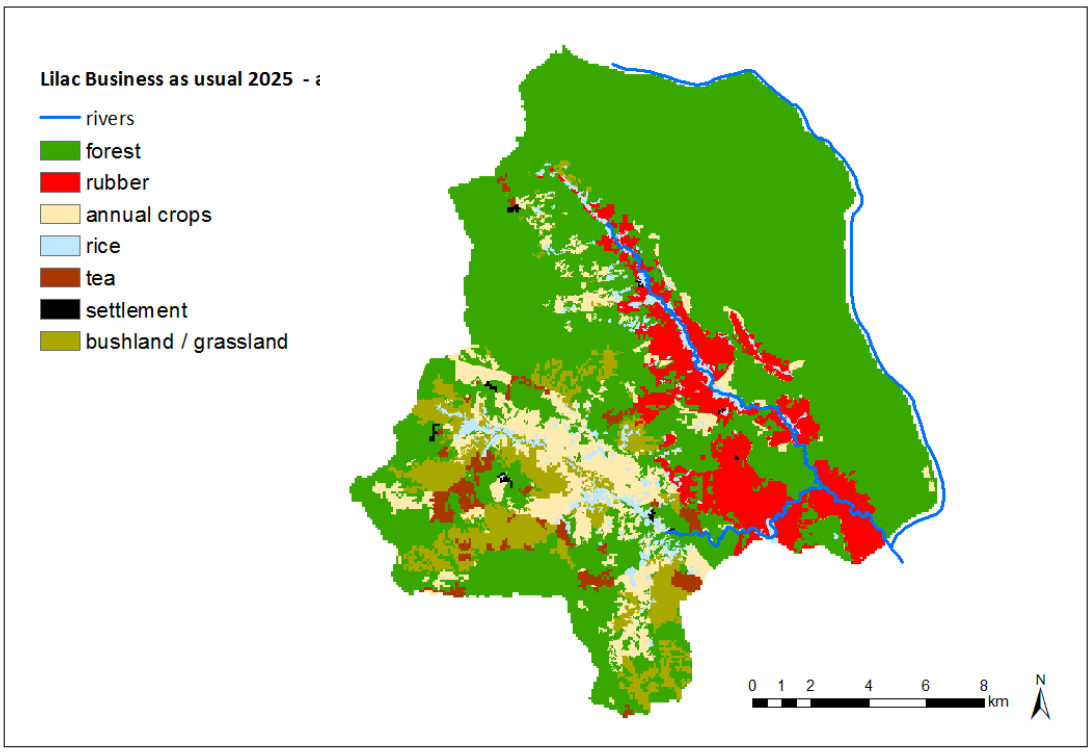

Figure D.3. Scenario 1, Business as usual

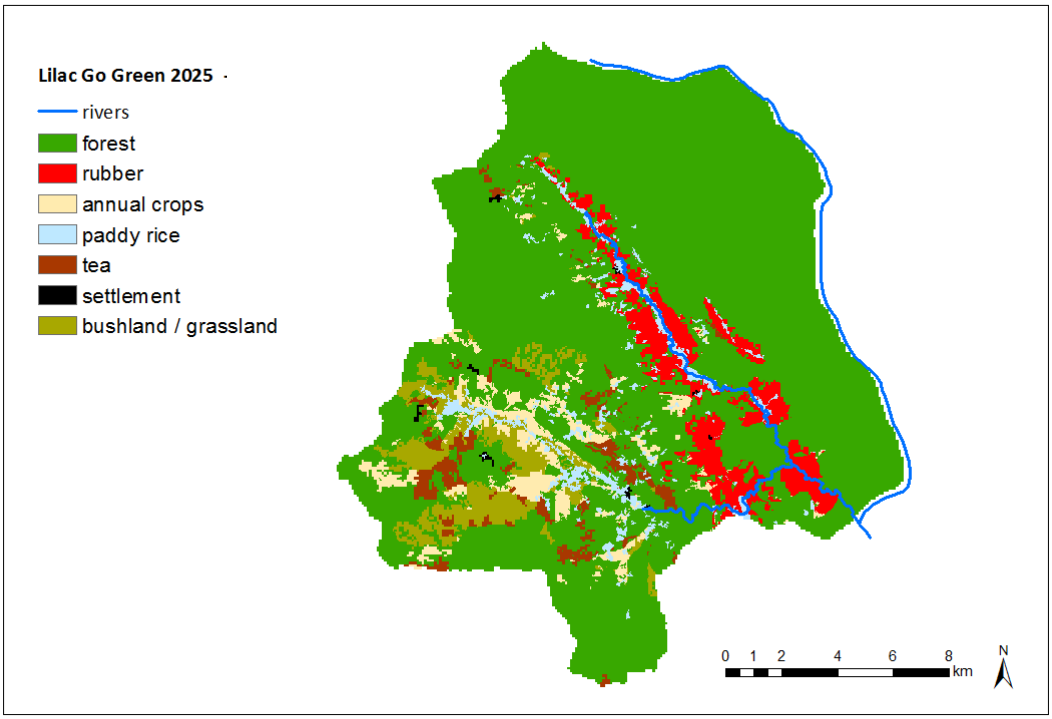

Figure D.4. Scenario 2, Go Green

Accessibility

Landscapes that are protected either legally or by geographic features can be included in the modeling process. Since the NRWNNR is managed according to the Man and Biosphere (MAB) Programme, the area is divided into a core zone, where all access is prohibited (input value 0), a buffer zone with limited access (input value 0.5) and an experimental zone with no restrictions (input value 1; Figure D5) (modified from Naban River Watershed National Nature Reserve Bureau 2014).

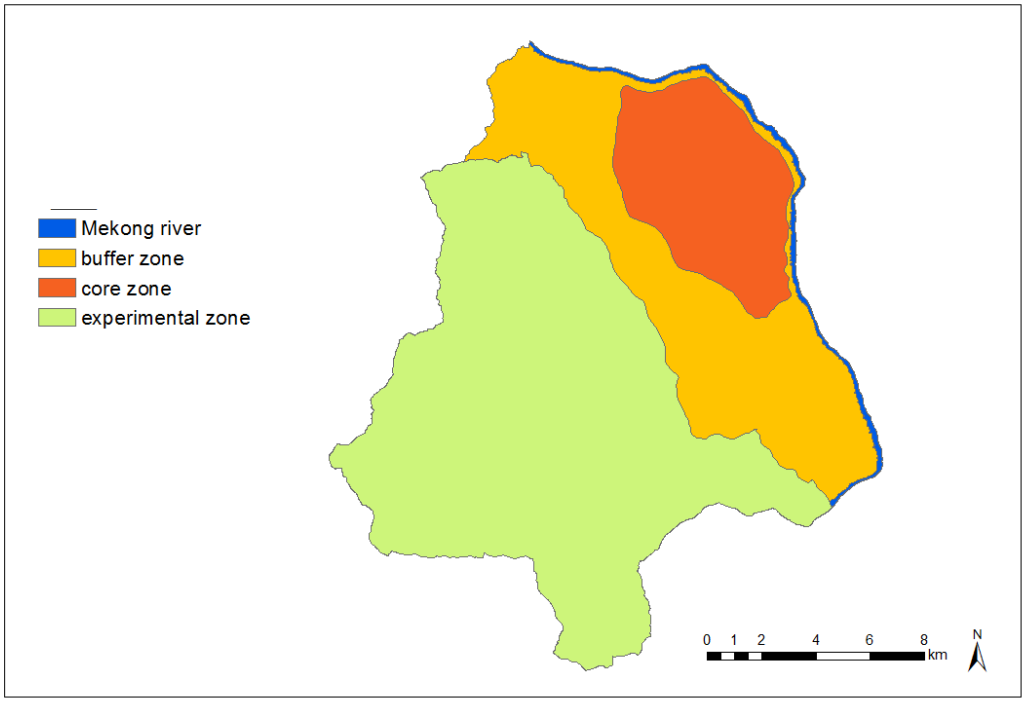

Figure D.5. Delineation of the different MAB zones in the research area

Habitat scores for land use classes

There are basically two options in InVest to model habitat quality. The first one is a simple consideration of which land use types are habitat or non-habitat in general, irrespective of different species. The underlying assumption is an island-ocean model, therefore, all natural habitats are considered as habitat and all managed land use classes as non-habitat. The second one is an assignment of relative habitat suitability scores to each land use. The underlying assumption is that species also "use" managed landscapes, although these might not be suited to provide all necessary functions (feed, suitability for reproduction) to allow occupation but they might be used temporarily. Since different species use different land use types as "intermediate" habitat, this approach is generally not meant for biodiversity in general, but for single species. We used a third approach for our analysis. From personal work, detailed data on presence/absence of different species from different groups (plants, vertebrates, invertebrates) in different land use classes are available. From these absolute data we derived relative habitat suitability scores for respective species groups. Thereby we are able to show how important different land use types are for different groups.

Since we did not have complete data for all land use classes (neither for our own data, nor from literature), we had to complete the missing data with values of similar land use classes (Table D.1).

Table D.1. Complete table with input habitat scores

land use class	red list habitat score	all species habitat score	human use habitat score
forest	1	1	1
rubber	0.36	0.57	0.82
annual crops	0.18	0.33	0.65
paddy rice	0.16	0.26	0.51
tea	0.12	0.28	0.49
settlement	0.1	0.1	0.1
bushland / grassland	0.32	0.38	0.56

Threats

Edge and fragmentation effects in InVEST are included by the inclusion of threats that can degrade a neighboring habitat. We included four threats into our analysis: three land use classes (rubber, annual crops and settlements) and roads. Threat data for the three land use class were extracted from the respective maps. The road map used includes major roads in the area (Figure D.6) (modified from Naban River Watershed National Nature Reserve Bureau 2014).

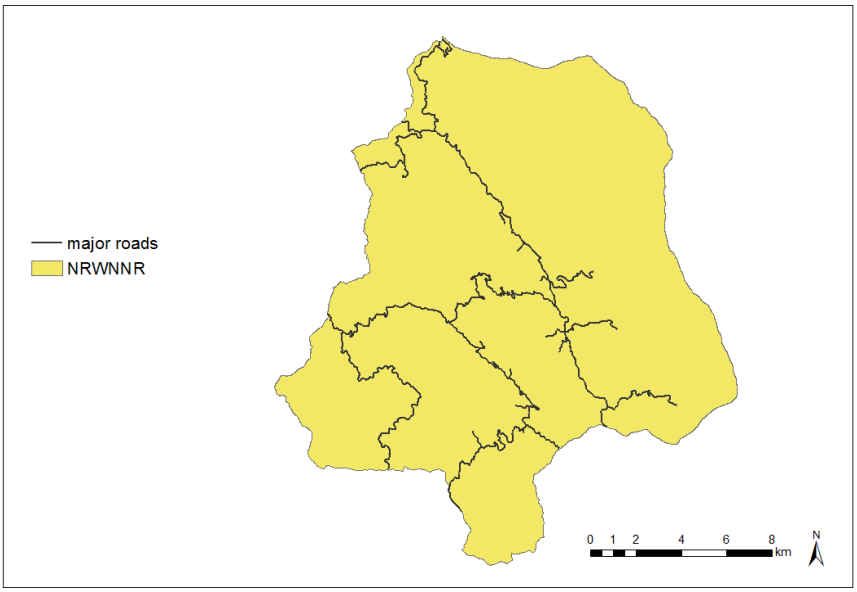

Figure D.6. Roadmap of the Naban River Watershed National Nature Reserve (NRWNNR)

For all threats three variables need to be defined: the maximum distance to what the threat has an effect (in km) (dist), if the threat effect decreases linearly or exponentially (decay), and how strong the effect is in relation to other threats (weight). Values for these variables were defined separately for the different species groups flora, vertebrates and invertebrates by expert panels. Values were averaged for the different species groups analyzed, all values for the aggregated biodiversity index (all species), the values of flora and vertebrates, for the red list species (red list) and the values of flora and invertebrates for the species with direct value for humans (human use). Table D.2 shows the final input values for the different model runs.

Table D.2. Values for threat data

	red list			all			human use		
Threat	**dist**	**decay**	**weight**	**dist**	**decay**	**weight**	**dist**	**decay**	**weight**
rubber	0.1	expon	0.35	0.1	expon	0.27	0.1	expon	0.3
annual crops	0.1	expon	0.3	0.1	expon	0.3	0.1	expon	0.3
settlements	1	expon	1	1	expon	1	1	expon	1
roads	0.1	expon	0.7	0.1	expon	0.5	0.1	expon	0.45

Sensitivity of habitats

The last requested input data are values on how sensitive the different habitats are to the threats rubber (rub), annual crops (ancr), settlements (settl) and roads (road). These values were also derived from expert panel meetings for the species groups flora, vertebrates and invertebrates and averaged (same procedure as for threat data) for the analyzed groups red list, all species and human use (Table D.3). Values close to 1 represent high sensitivity to a threat, whereas a value of 0 represents full insensitivity.

Table D.3. Values for sensitivity data

	red list				**all species**				**human use**			
land use class	**rub**	**ancr**	**settl**	**road**	**rub**	**ancr**	**settl**	**road**	**rub**	**ancr**	**settl**	**road**
forest	0.65	0.85	1	0.95	0.7	0.6	1	0.8	0.65	0.45	1	0.75
rubber	0	0.15	0.85	0.7	0	0.1	0.87	0.63	0	0.15	0.85	0.65
annual crops	0	0	0.4	0.4	0.1	0	0.5	0.33	0.15	0	0.35	0.2
paddy rice	0	0.05	0.4	0.4	0	0.07	0.47	0.33	0	0.05	0.3	0.2
tea	0	0	0.35	0.2	0	0	0.27	0.17	0	0	0.05	0.05
settlement	0	0	0	0	0	0	0	0	0	0	0	0
bushland grassland	0.3	0.5	0.65	0.5	0.23	0.33	0.5	0.4	0.05	0.15	0.25	0.25

Appendix E

Figure E.1. Examples of elephant pictures taken with camera traps in different locations in and around Tai Rom Yen National Park: At Khlong Hai (a), Phet Pha Nom Wat (b), Chong Chang (c) and around the Chulabhorn Development Project (d).

Figure E.2. Different ungulate species photographed at the forest boundary: Red muntjak (a), wild pigs (b), mousedeer (c) and serow (d).

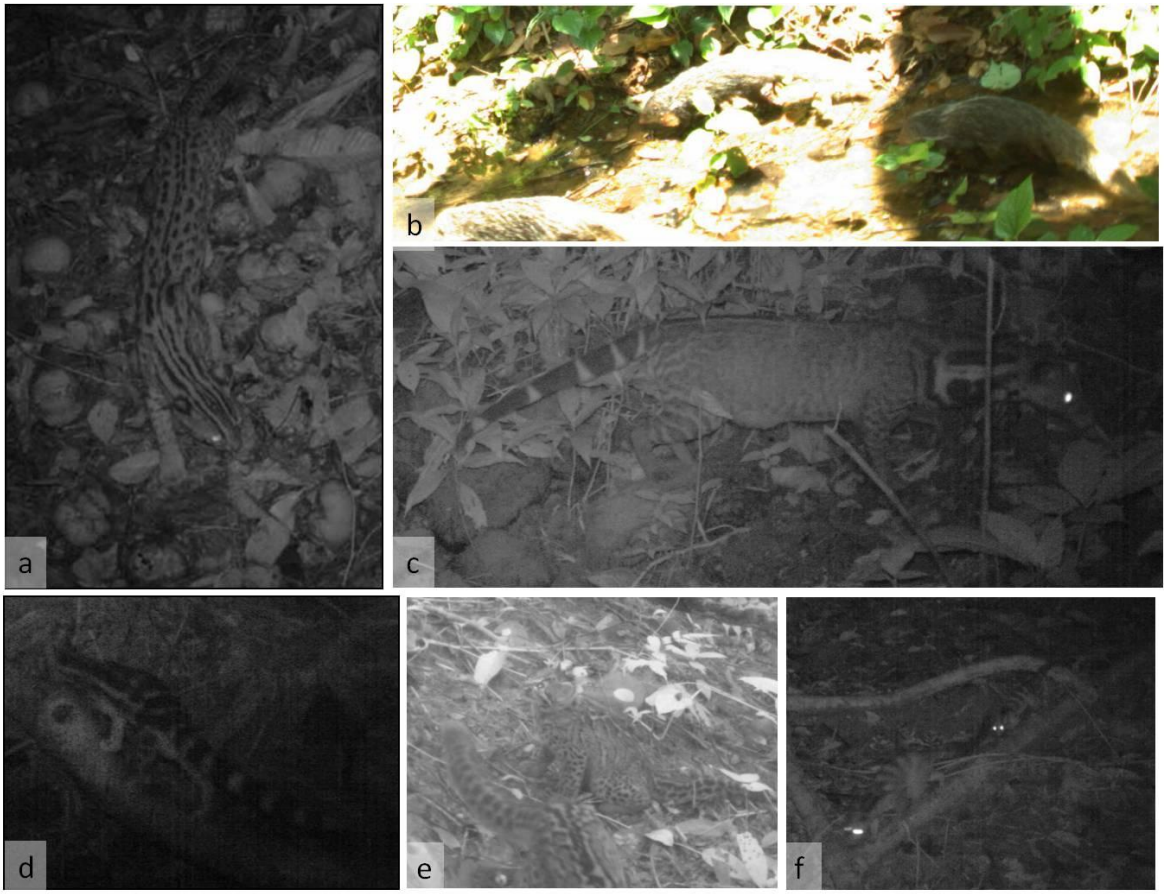

Figure E.3. Different carnivore species captured in the camera traps: Leopard cat (a), three crab-eating mongoose (b), large Indian civet (c), banded linsang (d), two marbled cats (e), two banded civets (f).

Figure E.4. Primate species recorded through the camera traps: Stump-tailed macaque (a), Southern pig-tailed macaque (b), and langur (c).

Appendix F

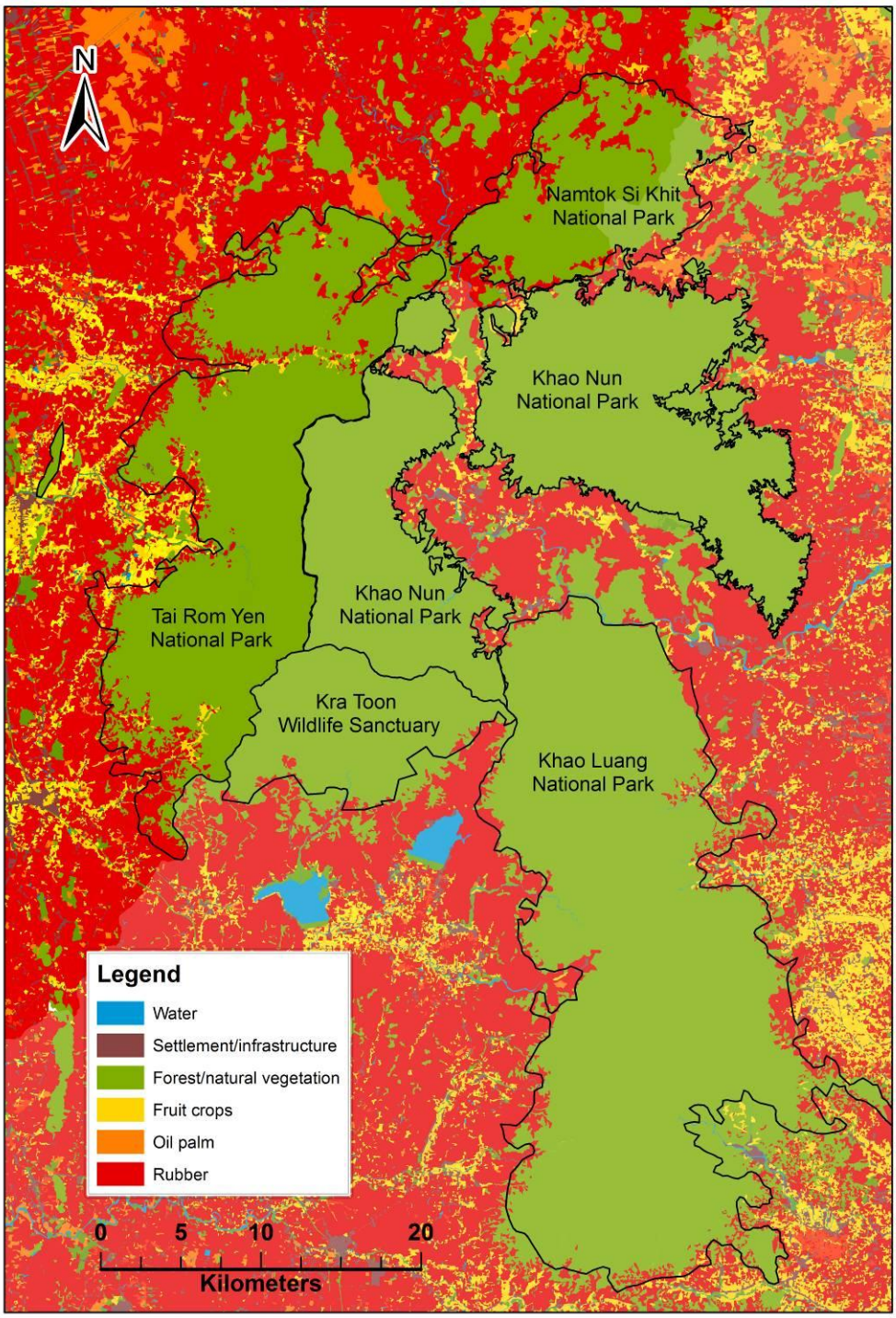

Figure F.1. Map of forest and conservation area complex to which Tai Rom Yen National Park belongs. Darker colors indicate the province of Surat Thani, paler colors represent the province of Nahkon Si Thammarat. The land-use map was provided by the Land Development Department Thailand.